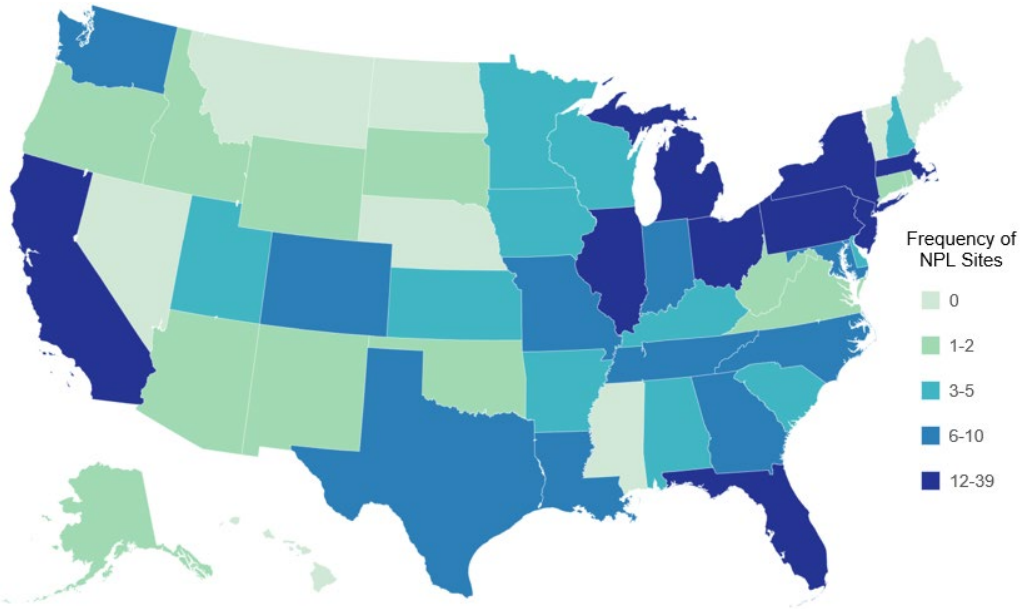
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

HCH isomers have been identified in at least 312 of the 1,867 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2019). However, the number of sites in which HCH isomers have been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Of these sites, 308 are located within the United States, 1 is located in the Virgin Islands, 1 is in Guam, and 2 are in Puerto Rico (not shown).

Figure 5-1. Number of NPL Sites with Hexachlorocyclohexane Contamination



Source: ATSDR 2019

- Registrations of γ-HCH agricultural products have been cancelled since 2006, significantly reducing most consumer exposure routes. The primary route of exposure is through medicinal use. One percent γ-HCH shampoos or lotions are registered with the FDA and are used for prescription treatment of lice and scabies.

- Individuals who live near contaminated sites may also experience higher exposures. Accidental ingestion or improper use of γ-HCH prescription treatments is likely the highest route of exposure.

- Historically, HCH has been released to the environment during its formulation process and through its use. HCH isomers are persistent and have been recently detected in air, water, and soil. The general public may be exposed to low levels of HCH through inhalation of contaminated ambient air, consumption of contaminated drinking water, or incidental ingestion of

or dermal contact with contaminated soils. HCH has not been found to be a major contaminant of drinking water supplies.

- Once released to the environment, HCH can partition to all environmental media. HCH can exist in the vapor and particulate phase in the atmosphere. HCH can volatilize from soils but is not expected to volatilize significantly from water. HCH has low to moderate mobility in soils and may leach to groundwater. HCH has low to moderate potential to bioaccumulate and has been detected in aquatic organisms in the United States.

- HCH has a long atmospheric lifetime but can be removed by photodegradation with hydroxyl radicals or wet and dry deposition. Biodegradation is believed to be the dominant decomposition process for HCH in soil and water, although hydrolysis and photolysis may occur to a lesser extent. The rates of degradation depend on the ambient environmental conditions.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Table 5-1 lists the facilities in each state that process γ-HCH, the intended use, and the range of maximum amounts of γ-HCH that are stored on site (TRI21 2022). Most of the uses by these facilities are considered to be ancillary, indicating purposes other than chemical processing or manufacturing. Examples of ancillary uses as defined under TRI include cleaners, degreasers, lubricants, fuels, and waste treatment uses.

Table 5-1. Facilities that Produce, Process, or Use γ-Hexachlorocyclohexane

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AR	1	1,000	9,999	9, 12
NE	1	10,000	99,999	9, 12
OH	2	1,000	99,999	12
TX	2	1,000	99,999	9, 12
UT	1	10,000	99,999	9, 12

[a]Post office state abbreviations used.
[b]Amounts on site reported by facilities in each state.
[c]Activities/Uses:

1. Produce
2. Import
3. Used Processing
4. Sale/Distribution
5. Byproduct
6. Reactant
7. Formulation Component
8. Article Component
9. Repackaging
10. Chemical Processing Aid
11. Manufacture Aid
12. Ancillary
13. Manufacture Impurity
14. Process Impurity

Source: TRI21 2022 (Data are from 2021)

HCH does not occur as a natural substance. The manufacturing of technical-grade HCH involves the photochlorination of benzene, which yields an isomeric mixture consisting of α-HCH, β-HCH, γ-HCH, δ-HCH, ε-HCH, and inerts (IARC 1979); this reaction can be started by free-radical initiators such as visual or ultraviolet light, X-rays, or γ-rays (Kirk and Othmer 1985). Treatment with methanol or acetic acid, followed by fractional crystallization, concentrates γ-HCH to the 99.9% required in the technical-grade of γ-HCH (IARC 1979); nitric acid is used to remove odor (NLM 2021). None of the isomers or technical-grade HCH are currently produced in the United States. The production of γ-HCH exceeded $2.27x10^6$ g in 1976 (NLM 2021); commercial γ-HCH production in the United States reportedly ended in that year (EPA 1989a).

γ-HCH was available in emulsifiable and flowable concentrates, soluble concentrates/liquids, wettable powders, dusts, ready-to-use liquids, pressurized liquids and impregnated materials, oil base and aerosol sprays, and granules, as well as a smoke generator (Berg 1988; EPA 1985a). γ-HCH was sold separately or in combination with fungicides, fertilizers, other insecticides, or wood preservatives (Hayes 1982).

5.2.2 Import/Export

Current data on the importation of γ-HCH were not located. γ-HCH was not included in import/export information submitted by manufacturers under EPA's Chemical Data Reporting (CDR) database in the 2016 (covering 2012–2015) or 2012 (covering 2011) reporting cycles (EPA 2012, 2016). Reporting thresholds were 2,500 and 25,000 pounds for the 2016 and 2012 cycles, respectively.

γ-HCH historically was imported from France, Germany, Spain, Japan, and China (EPA 1985a). Currently, India is the only country where γ-HCH is reportedly produced; thus, India may be the current supplier to the United States (Vijgen et al. 2011). The U.S. imports of γ-HCH declined from $1.52x10^5$ kg in 1977 to $8.53x10^4$ kg in 1982 (NLM 2021). In 2002, it was estimated that 90 metric tons ($9.0x10^4$ kg) of γ-HCH were imported into the United States (Hauzenberger et al. 2002). Up until 2001, it was estimated that 500 metric tons of γ-HCH-containing pesticide products were exported annually by the United States (primarily to Canada) (Hauzenberger et al. 2002). That export volume dropped to 25 metric tons in 2001 and continued to decline significantly as other countries and the United States ceased the usage of γ-HCH containing pesticides.

5.2.3 Use

γ-HCH was initially registered by the U.S. Department of Agriculture (USDA) in the 1940s and over the years, was approved for use on a wide variety of fruit and vegetable crops (including seed treatment), tobacco, greenhouse vegetables and ornamentals, forestry (including Christmas tree plantations), farm animal premises, and other uses. In February 1977, EPA issued a notice of Rebuttal Presumption Against Registration (RPAR), now called a Special Review, and continued registration of pesticide products containing γ-HCH. EPA took this action in response to indications of γ-HCH's potential carcinogenic effect, possible developmental and reproductive effects, possible blood dyscrasias, and delayed toxic effects, as well as its acute toxic effects seen in aquatic wildlife (IARC 1979). In October of 1983, EPA issued a "Notice of Intent to Cancel Pesticide Products Containing γ-HCH." The contentions concerning developmental and reproductive effects were successfully challenged by industry. EPA no longer permits the use of γ-HCH for purposes involving direct aerial application (EPA 1985b). The notice restricted certain applications of γ-HCH on livestock, structures, and domestic pets to certified applicators or persons under their direct supervision (EPA 1985b). In November 1993, EPA issued a "Notice of Receipt of a Request for Amendments to Delete Uses" for several formulations of γ-HCH powder, 99.5% technical-grade HCH, and dust concentrate, which would delete from the pesticide label most uses of γ-HCH for agricultural crops and use on animals and humans (EPA 1993). According to the EPA's last Registration Eligibility Decision (RED), the last approved food/feed use of γ-HCH that was supported for re-registration was seed treatment on barley, corn, oats, rye, sorghum, and wheat (EPA 2002). Since the 1998 and 1999 use deletions, the registrants were no longer interested in supporting the seed treatment use on broccoli, Brussel sprouts, celery, cabbage, cauliflower, collards, kale, kohlrabi, mustard greens, lettuce, radishes, spinach, or Swiss Chard (EPA 2002). Based on EPA estimates from 1996 to 2001, about 233,000 pounds of γ-HCH were used annually as a seed treatment (EPA 2002). In August 2006, EPA issued "Notice of Receipt of Requests to Voluntarily Cancel Lindane Pesticide Registrations," which would end the use of γ-HCH as seed treatments, and notice of final orders of cancellation was issued in December of 2006 (EPA 2006a, 2006b).

γ-HCH is available and regulated by the FDA, for the pharmaceutical treatment of scabies and head lice (EPA 2002). A 1% γ-HCH lotion is available for the treatment of scabies, and a 1% shampoo is available for the treatment of head lice; both are prescription use only. Both uses have been on the market since 1947, but they were labeled as a second line therapy in 1995 after a review by the FDA. The FDA has issued revised labels for 1% γ-HCH lotion and 1% γ-HCH shampoo to be used with caution for infants, children, and the elderly or anyone who weighs <110 pounds (50 kg), and people with other skin

conditions (FDA 2015). The products are contraindicated in premature infants and people with disorders that cause seizures (FDA 2015). In the past, γ-HCH was used in veterinary products to control mites and other pests, but recent data suggest that no products are currently registered in the United States for this use (Hauzenberger et al. 2002).

5.2.4 Disposal

HCH is listed as a toxic substance under Section 313 of the Emergency Planning and Community Right to Know Act (EPCRA) under Title III of the Superfund Amendments and Reauthorization Act (SARA) (EPA 2020c). Disposal of wastes containing HCH is controlled by a number of federal regulations.

The recommended disposal technique for γ-HCH is incineration, at 400–500 °C in the presence of a catalytic mixture of 5–10% metal chloride (copper, iron, zinc, or aluminum) on activated carbon (EPA 1975). Residence times based on this method were not reported. Other effective waste disposal methods include treatment with strongly alkaline solution or oxidation. In a laboratory-scale study, 98.5% γ-HCH was removed after a 6.5-hour treatment at pH 11.5 (EPA 1975). γ-HCH can be effectively oxidized by ozone and somewhat effectively oxidized by potassium permanganate; oxidation with chlorine or hydrogen peroxide was ineffective (EPA 1975). EPA standard treatment for hazardous wastes containing α-, β-, δ-, and γ-HCH is through either incineration or removal from liquid wastes by adsorption, prior to land disposal (EPA 2014).

While current disposal techniques may be adequate, new methods provide increased efficiency and quality of disposal at a greatly reduced cost. The use of demulsification, sorption, and filtration in combination with chemical and biological degradation of pesticide wastewaters is being examined. This process is divided into two phases. First, demulsification agents (lignocellulosic materials, peat moss, wood products, etc.) are utilized in the removal of solubilized pesticides. In the second phase, the solid matter (pesticide-saturated sorbents and suspended particulates) is physically separated from the aqueous material through a variety of filtration techniques. The aqueous phase is either recycled or discarded, and the solid phase, in which the concentration of the pesticide is most significant, is further treated through composting (Mullins et al. 1992).

In order to facilitate the composting process, it is important to use sorption agents that provide a beneficial environment for the pesticide-degrading microorganisms. Peat moss, ground pine bark mulch, and steam-exploded wood fibers are excellent demulsifiers because they are highly sorbent, readily

available, and inexpensive. They also provide the nutrients required by the degrading microorganisms, although the peat moss media require some carbohydrate enrichment. The solid waste can be either directly metabolized or co-metabolized by multiple species of microbes. The number of compost cycles, and therefore the amount of energy input required, depends on the pesticide concentration and on how easily the pesticide can be biodegraded. In preliminary studies by Mullins and coworkers, this process has reduced the concentration of γ-HCH in waste materials significantly, with <1% of the original pesticide remaining after 24-hour incubation (Mullins et al. 1992).

Additional work is required, but the benefits of this disposal technique are clear. It is cost-effective, reliable, and can be adapted to the variety of disposal challenges presented by the multitude of pesticides that are currently used. The use of microbial consortia ensures that each pesticide will be degraded rapidly. This method can also be used on pesticide mixtures (Mullins et al. 1992).

5.3 RELEASES TO THE ENVIRONMENT

The Toxics Release Inventory (TRI) data should be used with caution because only certain types of facilities are required to report (EPA 2005). This is not an exhaustive list. Manufacturing and processing facilities are required to report information to the TRI only if they employ ≥10 full-time employees; if their facility is included in Standard Industrial Classification (SIC) Codes 10 (except 1011, 1081, and 1094), 12 (except 1241), 20–39, 4911 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4931 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4939 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4953 (limited to facilities regulated under RCRA Subtitle C, 42 U.S.C. section 6921 et seq.), 5169, 5171, and 7389 (limited S.C. section 6921 et seq.), 5169, 5171, and 7389 (limited to facilities primarily engaged in solvents recovery services on a contract or fee basis); and if their facility produces, imports, or processes ≥25,000 pounds of any TRI chemical or otherwise uses >10,000 pounds of a TRI chemical in a calendar year (EPA 2005).

5.3.1 Air

Estimated releases of 189 pounds (~0.086 metric tons) of γ-HCH to the atmosphere from 7 domestic manufacturing and processing facilities in 2021 accounted for about 69% of the estimated total

environmental releases from facilities required to report to the TRI (TRI21 2022). These releases are summarized in Table 5-2.

Table 5-2. Releases to the Environment from Facilities that Produce, Process, or Use Hexachlorobenzenes[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
γ-HCH									
AR	1	0	0	0	35	0	0	35	35
NE	1	180	0	0	20	0	180	20	200
OH	2	0	0	0	1	0	0	1	1
TX	2	8	0	0	0	0	8	0	8
UT	1	0	0	0	0	28	0	28	28
Total	7	189	0	0	56	28	189	84	273

[a]The TRI data should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list. Data are rounded to nearest whole number.
[b]Data in TRI are maximum amounts released by each facility.
[c]Post office state abbreviations are used.
[d]Number of reporting facilities.
[e]The sum of fugitive and point source releases are included in releases to air by a given facility.
[f]Surface water discharges, wastewater treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).
[g]Class I wells, Class II-V wells, and underground injection.
[h]Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other onsite landfills, land treatment, surface impoundments, other land disposal, other landfills.
[i]Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown.
[j]The sum of all releases of the chemical to air, land, water, and underground injection wells.
[k]Total amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI21 2022 (Data are from 2021)

All isomers of HCH are considered hazardous air pollutants (HAPs) known to cause or suspected of causing cancer or other serious human health effects or harmful environmental effects (EPA 2020a), as regulated under the Clean Air Act. EPA's National Emission Inventory (NEI) database contains comprehensive and detailed estimates regarding sectors part of the emissions inventory system (EIS) which emit criteria air pollutants and HAPs for the 50 United States, Washington D.C., Puerto Rico, and the U.S. Virgin Islands. The NEI database includes point and non-point source emissions, onroad sources, non-road sources, and event sources such as emissions from wildfires or prescribed burning. According to data from the 2017 NEI, 62.40 pounds of γ-HCH were released from waste disposal,

industrial solvent use, industrial processes, and fuel combustion (EPA 2020b). These data are summarized in Table 5-3.

Table 5-3. HCH Emissions as Reported by the 2017 National Emission Inventory

Release sector	Emissions (pounds)
Waste disposal	54.607700078
Solvent; industrial surface coating and solvent use	5.91946
Industrial processes; ferrous metals	1.819
Industrial processes; NEC[a]	0.04
Fuel combustion; commercial/institutional oil	0.01

NEC = not elsewhere classified

Source: EPA 2020b

Historically, the largest source of γ-HCH releases to the air resulted from agricultural use of the pesticide γ-HCH, from aerial pesticide application or wind erosion of contaminated soils. γ-HCH may have also been released to the atmosphere via volatilization from treated agricultural soils and plant foliage (Lewis and Lee 1976). Evaporative loss of γ-HCH from water is not considered a significant potential source of atmospheric γ-HCH because of its relatively high water solubility (Mackay and Leinonen 1975). Quantitative historical estimates of the amount of γ-HCH released from these sources were not located in the literature. Aerial applications of γ-HCH were prohibited in the United States as its use as a pesticide was continuously restricted and eventually prohibited (EPA 1985b, 2006a, 2006b), and atmospheric releases from agricultural sources today are not expected.

5.3.2 Water

No releases of γ-HCH to water from 7 domestic manufacturing and processing facilities required to report to the TRI were reported in 2021 (TRI21 2022). This estimate includes releases to wastewater treatment and publicly owned treatment works (POTWs) (TRI21 2022). These releases are summarized in Table 5-2.

γ-HCH can be released to surface water via “down-the-drain” releases from consumer wash-off of treatments for lice and scabies (EPA 2002). These releases would be treated by wastewater treatment and POTWs, which did not report releases of γ-HCH to the TRI in 2019. Average γ-HCH concentrations in wastewater treated in Los Angeles County, California, have been reported to range from $3.6x10^{-5}$ to

0.03 µg/L (EPA 2002; Humphreys et al. 2008). α-HCH was not detected in 165 wastewater and wastewater treatment effluent samples collected between 2001 and 2010 (WQP 2021). One wastewater sample collected in 2003 contained a concentration of 0.014 µg/L α-HCH (WQP 2021). β-HCH was not detected in 24 wastewater samples collected between 2003 and 2009 (WQP 2021). Wastewater may be a negligible source of HCH to surface water and groundwater.

After soil or aerial application for agricultural use, γ-HCH could be released to surface water via surface runoff (as the dissolved chemical or adsorbed to particulates) or via wet deposition of rain and snow (Tanabe et al. 1982; Wheatley and Hardman 1965). Lake Ontario received 7 kg/year of α-HCH and <2 kg/year of γ-HCH due to suspended sediment loading from the Niagara River between 1979 and 1981 (Kuntz and Warry 1983). Historically, the Great Lakes in general received 0.77–3.3 metric tons/year of α-HCH and 3.7–15.9 metric tons/year of γ-HCH from atmospheric deposition of these contaminants (Eisenreich et al. 1981). Because γ-HCH is no longer allowed to be used for agricultural purposes, these are not expected to be significant sources of releases today.

Further from agricultural areas, urban stormwater runoff has historically resulted in releases of HCH to water in the range of parts per billion to parts per trillion. In 1982, α- and γ-HCH were detected in samples of urban stormwater runoff from Denver, Colorado, and Washington, D.C., at 0.0027–0.1 and 0.052–0.1 µg/L in 20 and 11%, respectively, of the 86 samples collected; β-HCH was detected only in runoff from Washington, D.C., in 5% of the samples at a concentration of 0.1 µg/L (Cole et al. 1984). In urban runoff samples collected in the Canadian Great Lakes Basin, γ-HCH was detected at mean concentrations of 0.0065 µg/L and 0.0035 mg/kg in the aqueous and sediment portions, respectively; the mean annual loading of the compound in runoff in the basin was reported to be 4.1 kg/year (Marsalek and Schroeter 1988). Stormwater samples collected in 2007, just after the ban on γ-HCH for agricultural use, showed concentrations between 6.0×10^{-5} and 0.14 µg/L α-HCH and 3.8×10^{-4} and 0.0976 µg/L β-HCH (WQP 2021).

γ-HCH could be released to groundwater via soil leachate after agricultural application or from improper disposal at contaminated sites. Available adsorption data indicate that γ-HCH has a low to moderate mobility in soils, and the results of monitoring studies suggest that γ-HCH does migrate to groundwater (Page 1981; Sandhu et al. 1978). Groundwater samples were collected from a packaging and reformulating pesticide facility in Florida, which had disposed of γ-HCH wastes in unlined trenches until 1996. Ground water concentrations for the site ranged from 30 to 420 µg/L for α-, γ-, and δ-HCH (Chartrand et al. 2015).

5.3.3 Soil

Estimated releases of 56 pounds (~0.025 metric tons) of γ-HCH to soil from 7 domestic manufacturing and processing facilities in 2021 accounted for about 20% of the estimated total environmental releases from facilities required to report to the TRI (TRI21 2022). No additional quantities were released via underground injection (TRI21 2022). These releases are summarized in Table 5-2.

γ-HCH has historically been released to the soil by direct application of the pesticide to soil, and can be released by direct or indirect releases during formulation, storage, and/or disposal. Hazardous waste sites where γ-HCH has been disposed of in the past are sources of γ-HCH in soils. However, the application of γ-HCH to laboratory refuse columns simulating municipal landfills indicated that γ-HCH did not volatilize or leach from the refuse surface, and movement through the column was slight, suggesting that codisposal of γ-HCH with municipal refuse will result in minimal releases (Reinhart and Pohland 1991; Reinhart et al. 1991).

5.4 ENVIRONMENTAL FATE

5.4.1 Transport and Partitioning

Air. HCH isomers are volatile, relatively persistent substances in the atmosphere and are expected to be capable of long-range transportation. HCH can exist in the vapor and particulate phases based on the reported vapor pressures of the isomers (NLM 2021). Volatilization of γ-HCH used as a seed treatment was confirmed, with 12–30% of the applied pesticide volatilizing within 6 weeks of planting the seed (Waite et al. 2001, 2007). Correspondingly, atmospheric concentrations of γ-HCH were variable and increased when pesticide usage occurred; α-HCH concentrations were less variable throughout the year (Hoff et al. 1992a). During the winter, higher ratios of α-HCH to γ-HCH reflected the movement of air containing the more persistent α-HCH isomer from the colder Arctic regions to the south, while the lower ratios in the summer reflected both increased γ-HCH usage in the region and the lack of movement of Arctic air (Hoff et al. 1992a). γ-HCH was also seen to move with warm air during the summer months from the lower United States (or areas even further to the south) to the Great Lakes region, although a similar trajectory could not be identified for the more ubiquitous α-HCH. Levels of α-HCH in air are not dominated by volatilization or partitioning to surfaces, but are dependent on local temperature changes (Hoff et al. 1992b). α-HCH appears to have a long residence time in the atmosphere and is controlled

primarily by transport. Long-range transport potentials were estimated for α- and γ-HCH based on North American monitoring data, and were reported to be 11,151 miles (17,946 km) and 6,047 miles (9,732 km), respectively (Shen et al. 2004). The potential for widespread global distribution has been reported in several studies (Hargrave et al. 1988; Knap and Binkley 1991; Tanabe et al. 1982; Wittlinger and Ballschmiter 1990).

HCH isomers in the atmosphere are likely to be subject to rain-out and dry deposition, which may result in the contamination of surface soil and water. γ-HCH removal rates were 2.5%/week by rainfall and 3.3%/week by dry deposition, and the estimated residence time of γ-HCH in the atmosphere was 17 weeks (Atkins and Eggleton 1971). The dry deposition flux rate of α-HCH ranged from 0.1 to 5.1 ng/m^2/day in deposition samples collected in June–August 1997 near the southern Baltic Sea (Wiberg et al. 2001). The flux rate of γ-HCH was 0.9–32.6 ng/m^2/day over the same time frame. Seasonal variation resulted in lower dry deposition rates during the winter months. In samples collected between February and March 1998, the flux rate for α-HCH ranged from 0.25 to 0.54 ng/m^2/day, and the dry deposition flux rate for γ-HCH was 3.4–14.1 ng/m^2/day (Wiberg et al. 2001). The dry deposition flux rate of γ-HCH in south central Saskatchewan in 1998 where it had been used as a seed treatment in a canola field ranged from <29 to 2,203 ng/m^2/day, and the amount in rainfall over the same period ranged from <10 to 200 ng/L (Waite et al. 2001). Uptake by plants may be another removal pathway, as observed for α- and γ-HCH under experimental conditions with lettuce, romaine, and garlic leaf (Yang et al. 2007). Removal was controlled by plant-air equilibration and correlated strongly with the reported log octanol-air partition coefficients (K_{OA}), 7.44 and 7.72 α- and γ-HCH, respectively (Yang et al. 2007).

Water. In surface waters, HCH has a slight tendency to dissolve and remain in the water column based on the water solubilities and octanol-water partition coefficients (K_{ow}) of the isomers (Clayton and Clayton 1981; Hansch and Leo 1995; Hollifield 1979; Kurihara et al. 1973). Although γ-HCH has a relatively high vapor pressure and Henry's law constant compared with many other organochlorine insecticides, evaporative loss of γ-HCH from water is not considered to be significant. Mackay and Leinonen (1975) calculated theoretical losses of several pesticides from saturated water solutions and predicted a volatilization half-life of 191 days for γ-HCH.

γ-HCH released to water may undergo adsorption/desorption with sediments and other materials in the water. Adsorption and desorption studies of γ-HCH in natural water-sediment systems performed by Saleh et al. (1982) indicate that a diversity of the natural water-sediment characteristics may affect the sorption-desorption behavior of γ-HCH in addition to the organic carbon content of the sediments.

γ-HCH is sorbed to silt solutions with a slow desorption rate, indicating that transport through the environment is most likely to be particle mediated (Noegrohati and Hammers 1992). Biosorption of γ-HCH was seen for the fungus *Rhizopus arrhizus* and activated sludge, with equilibrium being reached within 1 and 4 hours, respectively. Death of the sludge biomass resulted in rapid desorption with zero-order kinetics, suggesting that adsorbed γ-HCH can be released back into the environment (Tsezos and Wang 1991a). The sorption of γ-HCH from water using wood charcoal has been described (Keerthinarayana and Bandyopadhyay 1998); it was found to be a good sorbent for γ-HCH from water.

Sediment and Soil. HCH present in soil can leach to groundwater, sorb to soil particulates, or volatilize to the atmosphere. In general, the leaching of organic chemicals through soil is governed by the water solubility of the chemicals and their propensity to bind to soil. Based on the results of a number of laboratory soil column leaching studies that used soils of both high and low organic carbon content as well as municipal refuse, γ-HCH generally has low to moderate mobility in soils, with K_{oc} values ranging from 641 to 3,362; log K_{oc} range of 2.810–3.5266 (EPA 1982b; Melancon et al. 1986; Reinhart et al. 1991). Adsorption of γ-HCH to soil particulates is generally a more important partitioning process than leaching to groundwater. However, groundwater sediments, which have low organic carbon content (<0.1%), are not sufficient to adsorb γ-HCH to the extent that groundwater contamination is prevented (Nordmeyer et al. 1992). The presence of black carbon in soils from incomplete combustion may impact sorption affinity. HCH isomers showed varying preference for partitioning to black carbon (α-HCH > β-HCH > δ-HCH) in soils with 0.82–2.26% organic carbon and 0.04–0.5% black carbon (Ali et al. 2016). Sorption was observed to be a limiting factor in bioavailability of γ-HCH in soil to earthworms (Smídová et al. 2012).

Using sediment (0.44% organic carbon) obtained from a sugar-cane growing region of Australia, the K_{oc} of γ-HCH was measured as 2,164 (Just et al. 1990). The K_{oc} of γ-HCH in a mineral soil containing 1.26% organic carbon content was measured as 832 (Chiou et al. 1998). In a sandy soil (0.105% organic carbon) γ-HCH had a measured K_{oc} of 3,362, and a desorption K_d of 3.53 (Melancon et al. 1986). The partition coefficient (K_p) of γ-HCH in a laboratory column experiment with municipal solid waste was 853 (Reinhart et al. 1991). In a study involving a laboratory sediment/water system (pH 7.42; 2.18% organic carbon), α- and γ-HCH isomers were adsorbed on sediments under both aerobic and anaerobic conditions and few differences were noted in the adsorption behavior of each isomer. Under aerobic and anaerobic conditions, the K_{oc} values of α-HCH were 681 and 617, respectively, while the K_{oc} values for γ-HCH were 641 and 694, respectively (Wu et al. 1997). A mixture of HCH isomers (α-, β-, γ-, and δ-HCH) sorbed very strongly to a soil from Nagpur, India (pH 7.6, 0.387% organic carbon), with a K_{oc}

value of 54,000. Some desorption was observed, believed to be due to the water solubility of HCH (Wadaskar et al. 2006). Desorption experiments with a sandy loam soil slurry showed isomeric differences in desorption capacity, α- ≥ γ- > δ- > β-HCH (Quintero et al. 2005).

γ-HCH sorbed to the soil can partition to the atmosphere by wind erosion of surface soil particulates (Stanley et al. 1971) and via volatilization from treated agricultural soils and plant foliage (Lewis and Lee 1976). In tests conducted in a model laboratory system at 10 and 20°C, volatilization half-lives of γ-HCH from soil and oat plant surfaces of 2.3–24.8 and 0.29–0.73 days, respectively, were reported (Dorfler et al. 1991a); half-lives were greater on dry, sandy soils versus peat soils; however, when moisture was added to the soils, the half-life was greater for the peat soil, while warmer temperatures decreased the half-life under all soil and moisture conditions (Dorfler et al. 1991b). In tests performed with a wind tunnel, a volatilization rate of >20% for γ-HCH from soil surfaces within a 24-hour period was determined (Rüdel 1997). A 6-fold increase in γ-HCH volatilization from soil was seen in laboratory experiments when the temperature increased from 15 to 45°C; flooding the soil also increased the volatilization (Samuel and Pillai 1990). A field study conducted in south central Saskatchewan, Canada in 1997–1998 in which γ-HCH was applied as a seed treatment to canola, determined that between 12 and 30% of the initial amount applied volatilized to the atmosphere (Waite et al. 2001); a follow-up study determined volatilization rates of 190 mg/hectare/week at 1 week after application and 420 mg/hectare/week 2 weeks after application (Waite et al. 2007). The volatilization rate from plant surfaces was 55% for γ-HCH. Application of γ-HCH to fields of sunflowers and sugarbeets resulted in a 54% evaporative loss of the pesticide within 24 hours (Neururer and Womastek 1991).

Other Media. γ-HCH has a low to moderate potential to bioaccumulate. A summary of some of the bioconcentration factors (BCFs) from experimental studies with γ-HCH are provided in Table 5-4. γ-HCH shows limited uptake from soils and bioconcentration by plants and terrestrial organisms and does not appear to biomagnify to a great extent.

Table 5-4. Results of Experimental Bioaccumulation Studies with γ-Hexachlorocyclohexane

Species	Exposure route	Bioconcentration factor	Reference
Brine shrimp	Water	183	Matsumura and Benezet 1973
Rainbow trout (fry)	Water	319	Ramamoorthy 1985
Pink shrimp	Water	84	Schimmel et al. 1977
Pinfish	Water	218	
Grass shrimp	Water	63	
Sheepshead minnow	Water	490	
Brine shrimp	Sand	95	Matsumura and Benezet 1973
Northern brook silverside fish	Sand	1,613	

A BCF of 1,273 (lipid basis) in prawns (crustacean) was seen to be 0.58 times the γ-HCH concentration in the underlying sediment, indicating that aquatic organisms may accumulate γ-HCH from the water column, and uptake from contaminated sediment alone may not be extensive (Just et al. 1990). BCFs for the isomers of HCH, using zebra fish under steady-state conditions, were 1,100 for α-HCH, 1,460 for β-HCH, 850 for γ-HCH, and 1,770 for δ-HCH; BCFs determined by uptake and clearance rate constants were slightly lower (Butte et al. 1991). Elimination of γ-HCH occurred rapidly in zebra mussels (BCF of 10) and metabolism of γ-HCH was not observed (Berny et al. 2002). BCFs on a wet weight basis for γ-HCH in different fish species were positively correlated with their lipid content (Geyer et al. 1997). The bioaccumulation of γ-HCH by tubificide oligochaetes from a static system consisting of sediment and water has been reported (Egeler et al. 1997). Microalgae *Scenedesmus quadricauda* and *Coccomyxa subellipsoidea* were exposed for 24 hours to mine dump effluent containing α-, β-, γ-, and δ-HCH, resulting in bioaccumulation factors (BAFs) of 74.6, 60.5, 29.4, and 107.2 for α-, β-, γ-, and δ-HCH, respectively, in *S. quadricauda,* and BAFs of 50.8, 47.6, 21.5, and 56.3 for α-, β-, γ-, and δ-HCH, respectively, in *C. subellipsoidea* (Kováčik et al. 2018).

γ-HCH applied to an aquatic mesocosm (i.e., a small, artificial ecosystem) at 61.3 µg/L was reduced by 50% at 24 hours post-application, while at 19 weeks post-application, the concentration in the water was only 0.2%; no γ-HCH was detected at 21 weeks. The biological half-life was estimated to be 16.7 days. Movement through the water column was shown by increasing sediment concentrations up to a maximum of 75.4 µg/kg at 96 hours post-application; however, sediment concentrations decreased to below the detection limit at 23 weeks to give a half-life in sediment of 48.1 days. Rooted aquatic macrophytes have a BCF of 56 at a maximum concentration of 1.7 mg/kg at 24 hours post-application; however, at 14 weeks, all residues were below the detection limit for a half-disappearance time of 18 days.

Gastropods in the system had a maximum γ-HCH concentration of 7.2 mg/kg at 24 hours post-treatment, yielding a BCF of 232.4 and a half-disappearance time of 13.7 days with all residues eliminated by 13 weeks (Caquet et al. 1992).

Trophic transfer of γ-HCH may occur in the foodweb. A study assessed the potential of transfer between zebra mussels and their predators, tufted ducks (*Aythia fugigula*) at the birds' wintering grounds in Lake Geneva, bordering Switzerland and France. γ-HCH concentrations in the mussels ranged from approximately 5 to 500 ng/g wet weight, while γ-HCH in the liver of the tufted ducks ranged from 8.9 to 175.6 ng/g wet weight (Bemy et al. 2003).

In tests with radiolabeled γ-HCH, grain, maize, and rice plants accumulated 0.95, 0.11, and 0.04%, respectively, of the amount of bound residues following 14–20 days growth in a sandy loam soil. Bioconcentration increased by 4–10 times when the plants were grown in test soils containing both bound and extractable residues of γ-HCH (Verma and Pillai 1991). Plants and grains grown on soil treated with γ-HCH showed α-HCH as the predominant isomer, although all isomers were found to some extent; amounts decreased with increasing time after application (Singh et al. 1991). A different trend in isomer uptake was observed in garlic. Garlic (*Allium sativum* L.) was planted in pots containing soil treated with α-, β-, γ-, and δ-HCH isomers. The BCFs of the underground parts were in the range of 0.48–0.90, 1.60–1.84, 1.40–2.34, and 2.60–3.64 for α-, β-, γ-, and δ-HCH, respectively, and above-ground parts were 1.50–2.26, 4.50–6.79, 5.16–6.81, and 9.30–12.18, respectively (Chen et al. 2013). The phytoavailability of the isomers was observed to be δ- > γ- ≥ β- > α-HCH, which generally agreed with the isomer's water solubility and vapor pressure (Chen et al. 2013). Evidence of plant uptake from air has been reported. Lettuce, romaine, and garlic leaf were maintained in air chambers that exposed them to air polluted with α- and γ-HCH for 5 days. Measured accumulation factors of α-HCH in the crops were 77.25, 190.8, and 95.23 for lettuce, romaine, and garlic leaf, respectively, and accumulation factors of γ-HCH were 187.6, 321.9, and 124.5, respectively (Yang et al. 2007).

Uptake of γ-HCH by earthworms from a treated soil has also been reported. *Eisenia andrei* exposed to grassland soil spiked with γ-HCH for 24 hours had measured BAFs ranging from 5 to 35 (Šmídová et al. 2015). Following exposure to 5 ppm of the compound for up to 8 weeks, earthworms bioconcentrated γ-HCH by a factor of 2.5. The earthworms biotransformed more than 50% of the accumulated γ-HCH; the main degradation product was γ-2,3,4,5,6-pentachlorocyclohex-1-ene (Viswanathan et al. 1988).

γ-HCH and the other isomers of HCH do not appear to undergo biomagnification in terrestrial food chains to a great extent, although there is a moderate potential for transfer of γ-HCH to animal tissue as a result of soil ingestion or ingestion of contaminated foliage (Wild and Jones 1992). Clark et al. (1974) found that γ-HCH levels in the adipose tissue of cattle were 10 times higher than in the feed (0.002 mg/kg). Szokolay et al. (1977) examined relative accumulation of HCH isomers including γ-HCH and various components in the food chain in Czechoslovakia. Lower γ-HCH residues were found in tissues of animals (chickens, sheep, pigeons) feeding entirely on plant material, whereas carnivores had higher concentrations.

γ-HCH that is adsorbed to sediments may be recycled to the atmosphere as gas bubbles are formed in the sediment by the methanogenesis and denitrification processes of bacteria. In one case studied, it was estimated that 85% of the γ-HCH associated with the sediment gas bubbles would be released to the atmosphere, with the remaining 15% being dissolved in the water column as the bubble rises toward the surface (Fendinger et al. 1992).

5.4.2 Transformation and Degradation

Air. HCH is degraded in the atmosphere by reacting with photochemically produced hydroxyl radicals. The rate of this reaction is not very rapid however, and all of the HCH isomers have rather long atmospheric lifetimes. The rate constants for the reaction of α- and γ-HCH with hydroxyl radicals were measured as 1.4×10^{-13} and 1.9×10^{-13} cm^3/molecule-second, respectively (Brubaker and Hites 1998). Using an average hydroxyl radical concentration of 5×10^5 molecule/cm^3, the corresponding half-lives are about 115 and 84 days for α- and γ-HCH, respectively. In locations where the atmospheric hydroxyl radical concentration is very low, the persistence times of these compounds are much longer. Cortes and Hites (2000) estimated that the average half-life of α- and γ-HCH around the Great Lakes region ranged from about 3 to 4 years. Since HCH does not absorb light >290 nm, direct photolysis in the atmosphere is not expected to be an important environmental fate process. However, Chen et al. (1984) reported photodegradation half-lives of 91, 152, 104, and 154 hours for thin films of α-HCH, β-HCH, γ-HCH, and δ-HCH, respectively, when irradiated with light of wavelength 295–305 nm. No absorption bands were observed in this spectral region, however, for any of the HCH isomers, and the mechanism of photodegradation and its environmental significance are uncertain. A direct photolysis study of α-HCH showed maximum absorption in the middle ultraviolet (UV) range at 252 nm, with a half-life around 2 hours (pseudo-first-order rate constant of 0.34/hour) (Zhang et al. 2014). The environmental relevance of this is unclear, since the middle UV range wavelengths are filtered out by the stratosphere. Similar

indirect photolysis studies were conducted at ≥280 nm with α-HCH in the presence of H_2O_2 at a molar ratio of 100:1 (H_2O_2:α-HCH). The indirect photolysis half-life was around 4.3 hours (pseudo-first-order rate constant 0.16/hour). A proposed final photolysis product is 2,4,6-trichlorophenol (Zhang et al. 2014).

Water. Biodegradation is believed to be the dominant degradative process for γ-HCH in aquatic systems, although hydrolysis and indirect photolysis may also occur. Sharom et al. (1980) found that <30% of the applied γ-HCH remained in unsterilized natural waters in capped bottles after 16 weeks. Biodegradation was concluded to be responsible for these results, although it was unclear to what extent hydrolysis or adsorption to the glass bottles may have contributed. Zoeteman et al. (1980) estimated river, lake, and groundwater half-lives for γ-HCH from degradation data in these environments to be 3–30, 30–300, and >300 days, respectively. In natural lake water with a pH of 9.0 and a hardness of >600 mg calcium carbonate/L, the half-life of γ-HCH was estimated to be 65 hours (Ferrando et al. 1992). γ-HCH, applied at concentrations of 50 or 500 μg/L to aerobic batch cultures of microorganisms with sodium acetate as a carbon source, was initially removed by adsorption and followed by desorption onto the biomass with subsequent decomposition (McTernan and Pereira 1991). Approximately 56–62% of the γ-HCH was removed from the water column in 23 days, with 26% removal by adsorption onto the biological solids produced in these batch reactors. Microbial growth, using γ-HCH in the absence of sodium acetate, increased as the microorganisms became acclimated; the pesticide still showed toxic properties, as evidenced by a concurrent increase in microbial death rates. Evidence of biodegradation of HCH isomers in groundwater has also been reported. In an *in situ* study of a former pesticide formulating plant, the biodegradation half-lives of α-, β-, and δ-HCH isomers were determined in groundwater below the site by compound-specific stable carbon isotope analysis respectively. Half-lives were determined based on isotopic depletion from samples collected over 3 years at various wells spreading out from the contaminant source. Half-lives were 223, 62–287, and 120–632 days for α-, β-, and δ-HCH isomers, respectively (Bashir et al. 2015).

It has been shown that γ-HCH is degraded by nitrogen-fixing blue-green algae. These algae reduce the toxic effects of γ-HCH following repeated inoculations (Kar and Singh 1979b). The degradation of γ-HCH became more efficient with time, thus reducing the pesticide's toxicity in cultures of nitrogen-fixing blue-green algae. Dechlorination of γ-HCH to γ-pentachlorocyclohexene was also shown to occur with fungi in aqueous suspensions (Macholz and Kujawa 1985) and in algal cultures (Sweeney 1969).

Hydrolysis is not considered an important degradation process for HCH in aquatic environments under neutral pH conditions. However, under alkaline conditions, γ-HCH is hydrolyzed fairly rapidly. Saleh et

al. (1982) tested rates of hydrolysis of γ-HCH in sterilized natural waters at 25°C and found that hydrolysis of γ-HCH followed first-order kinetics with half-lives of 92 hours at pH 9.3, 648 hours at pH 7.8, and 771 hours at pH 7.3. EPA (1989b) reported a hydrolysis half-life of 207 days at pH 7 and 25°C using distilled water. Alkaline hydrolysis (pH 9.78) of α-HCH was observed with a calculated half-life of 1,083 hours (based on pseudo-first-order rate constant of 0.0064/hour), giving 1,3,4,5,6-pentachlorocyclohexane, 1,2,4-trichlorobenzene, and 1,2,3-trichlorobenzene as the major products (Zhang et al. 2014).

Somewhat conflicting information is available on the rate of photolysis of γ-HCH in water. Since HCH does not contain chromophores that absorb light >290 nm, direct photolysis is not expected to occur. However indirect photolysis, whereby a photosensitizing agent may absorb light and then transfer its excitation energy to HCH, may occur. Humic and fulvic acids are well-known photosensitizing agents and are practically ubiquitous in natural waters. In the study by Saleh et al. (1982), the authors reported γ-HCH first-order photolysis half-lives of 169, 1,791, and 1,540 hours in pond water, lake water, and water from a quarry at pH 9.3, 7.3, and 7.8, respectively, when solutions were exposed to direct sunlight. However, the rapid rate of degradation at pH 9.3 may have been enhanced by hydrolysis reactions rather than by photolysis. In another study, α- and γ-HCH were shown to undergo enhanced photolysis when aqueous solutions were spiked with 5 and 25 ppm of soil fulvic acid, and irradiated with natural sunlight (Malaiyandi et al. 1982). Hamada et al. (1981) found that γ-HCH underwent photodegradation to form two isomers of tetrachlorohexene and pentachlorohexene in propanol solution when irradiated with UV light produced by a low-pressure mercury lamp. Oxidants commonly found in natural waters, such as peroxy radicals, hydroxyl radicals, and singlet oxygen species, can degrade HCH in water. Mill (1999) estimated that the indirect photolysis half-life of HCH in natural waters is about 270 days, and the dominant oxidant for HCH was the hydroxyl radical. Photolysis of γ-HCH in aqueous solution in the presence of polyoxomethallate, a strong oxidizing agent, has also been demonstrated (Hiskia et al. 1997).

Sediment and Soil. γ-HCH in soil or sediment is degraded primarily by biodegradation, although hydrolysis may occur in moist soils under alkaline conditions. Tu (1976) reported that 71 of 147 microorganisms isolated from a loamy sand soil were able to utilize a γ-HCH solution as the sole carbon source. White rot fungus degraded radiolabeled γ-HCH in aerobic pure culture laboratory tests. In a silt loam soil/corncob test matrix, 34.7% of the compound was degraded over a 60-day test period, whereas 53.5% degradation was observed in liquid cultures over a 30-day test period (Kennedy et al. 1990). The results of this study have been confirmed by more recent studies (Mougin et al. 1996, 1997). The isolation of γ-HCH-degrading bacteria, classified as *Sphingomonas paucimobilis*, from contaminated

soils has been reported (Thomas et al. 1996). A *Pseudomonas* species has also been isolated from pretreated soil that is able to degrade γ- and α-HCH, but not β-HCH, within 10–20 days under both flooded (anaerobic) and unflooded (aerobic) conditions; greater degradation rates were observed under aerobic conditions (Sahu et al. 1993). Under aerobic conditions, actinobacteria strains of *Streptomyces* sp. isolated from a polluted site were able to utilize α-HCH as its only carbon source, and some were able to utilize β-HCH when supplemented to the α-HCH system; no growth was observed in the presence of δ-HCH. At pH 7 and 30°C, the actinobacteria were able to degrade up to 100% α-HCH and 55% β-HCH after 7 days (Sineli et al. 2014). The concentrations and persistence of γ-HCH in soil may be dependent on soil types. An analysis of two soil types, loamy sand (approximately 1–2% organic matter) and muck (approximately 27–56% organic matter), for γ-HCH residues showed that mean residues in the loamy sand soil had decreased from 95 ppb dry weight in 1971 to below the detection limit of 10 ppb in 1989; however, in muck, residues had decreased from 426 ppb in 1971 to 168 ppb in 1989 (Szeto and Price 1991). The presence of crops on the soils also affects the persistence of HCH residues, with half-lives of 58.8 and 83.8 days for cropped and uncropped plots, respectively. β-HCH was the most persistent isomer, with half-lives of 184 and 100 days, respectively, on cropped and uncropped plots; γ-HCH was next at 107 and 62.1 days, followed by α-HCH at 54.4 and 56.1 days, and finally, δ-HCH at 33.9 and 23.4 days. Only trace amounts of the isomers were found to leach below 20 cm soil depth (Singh et al. 1991). The β-HCH isomer comprised 80–100% of the total HCH residues found in soil or vegetation on land surrounding an industrial landfill in Germany 10 years after the final HCH input (Heinisch et al. 1993). Biodegradation was observed to be a limiting factor in uptake of γ-HCH by earthworms (Smídová et al. 2012).

Most available information suggests that γ-HCH transformation is favored in biologically rich, anaerobic environments (EPA 1979; Haider 1979; Kalsch et al. 1998). In bench-scale anaerobic digestion tests designed to assess the fate of semivolatile organic pollutants in primary and secondary sludges, γ-HCH was found to undergo 98% degradation at 120 days. Sorption of the compound to the digester solids accounted for 2% of the initial feed; none of the compound was lost by volatilization. The digesters were operated at 35°C with a 30-day solids retention time (Govind et al. 1991). Similar results were seen with live activated sludge where initially reversible biosorption dominates the removal process followed by an increased aerobic biodegradation after approximately 10 hours of acclimation. The biodegradation process includes hydrolytic dechlorination with subsequent ring cleavage and finally, partial or total mineralization (Tsezos and Wang 1991b). Adaptation of sewage sludge is slow and may take 1–2 months; however, once acclimation occurs, 70–80% biodegradation of γ-HCH may occur, with the percentage of degradation decreasing with increasing sludge age (Nyholm et al. 1992). Co-oxidation and

reductive dechlorination are the probable degradation mechanisms (Jacobsen et al. 1991; Nyholm et al. 1992).

Numerous diverse studies on biological degradation have shown that γ-HCH was transformed to tetrachlorohexene; tri-, tetra-, and pentachlorinated benzenes; penta- and tetracyclohexanes; other isomers of HCH; and other related chemicals. The products varied depending on the organisms present, analytical methods applied, and when the sample was analyzed relative to its collection date (EPA 1979). Laboratory studies have demonstrated the bioisomerization of γ-HCH to α-, β-, and δ-HCH but bioisomerization in the environment was considered to be nonsignificant by an investigator who conducted a field study (Waliszewski 1993). Levels of individual isomers were approximately 0.1–1.4 and 0.8–4.0% of the γ-HCH concentrations at 3–31 and 34–46 weeks, respectively, following γ-HCH treatment of soil. The study authors suggested that their inability to simulate all environmental conditions in the laboratory could explain differences between laboratory and field results.

Abiotic transformation and degradation processes of γ-HCH in soil/sediment are not thought to be significant pathways. As discussed earlier for water, photolysis or hydrolysis are not considered important degradation pathways of γ-HCH and other isomers; the exception being hydrolysis under alkaline conditions.

Other Media. Several Organisation for Economic Cooperation and Development (OECD) and European Union standardized tests exist to quantify potential for biodegradation in a wastewater treatment facility. A closed bottle test, conducted according to EC directive 92/69/EEC, was initiated with 2 mg/L of γ-HCH in a 25 mg/L slurry of activated sludge in mineral nutrient medium, under aerobic conditions. γ-HCH achieved 100% degradation based on theoretical oxygen uptake after 9 days (Lapertot and Pulgarin 2006).

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to HCH depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of HCH in unpolluted atmospheres and in pristine surface waters are often so low as to be near the limits of current analytical methods. In reviewing data on HCH levels monitored or estimated in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable.

Table 5-5 shows the lowest limit of detections that are achieved by analytical analysis in environmental media. An overview summary of the range of concentrations detected in environmental media, including historical data prior to the cancellation of γ-HCH as a pesticide, is presented in Table 5-6.

Table 5-5. Lowest Limit of Detection Based on Standards[a]

Media	Detection limit	Reference
Air	0.2 pg/m^3 – 200 ng/m^3 (α-, β-, γ-HCH)	EPA 1999d
Drinking water	0.0053 µg/L (α-HCH) 0.0036 µg/L (β-HCH) 0.0060 µg/L (γ-HCH) 0.0020 µg/L (δ-HCH)	EPA 1995
Surface water and groundwater	7 pg/L – 0.0053 µg/L (α-HCH) 6 pg/L – 0.0036 µg/L (β-HCH) 9 pg/L – 0.0060 µg/L (γ-HCH) 5 pg/L – 0.0020 µg/L (δ-HCH)	EPA 1995, 2007
Soil	6 ng/L; 1.3 ng/kg (α-HCH) 7 ng/L; 0.6 ng/kg (β-HCH) 11 ng/L; 0.7 ng/kg (γ-HCH) 5 ng/L; 2.0 ng/kg (δ-HCH)	EPA 2000b, 2007
Sediment	0.500 µg/kg; 6 ng/L (α-HCH) 0.221; 7 ng/L (β-HCH) 0.200 µg/kg; 11 ng/L (γ-HCH) 5 ng/L (δ-HCH)	EPA 2000b; USGS 2003
Whole blood	1 ppb (α-, β-, γ-HCH) 1.3 ng/g lipid (β-HCH) 0.92 ng/g lipid (γ-HCH)	CDC 2019; EPA 1980

[a]Detection limits based on using appropriate preparation and analytics. These limits may not be possible in all situations.

Table 5-6. Summary of Ambient Environmental Levels of HCH

Media	Low	High	Reference
α-HCH			
Outdoor air (ng/m^3)	0.00016	0.142	Goel et al. 2010; WQP 2021
Indoor air	No data		
Surface water (ng/L)	1.4x10^{-5}	55,000	WQP 2021
Groundwater (ng/L)	0.96	2.5x10^6	WQP 2021
Drinking water	No data		
Food (ppm)		0.0010	Rogers et al. 1995
Soil and sediment (µg/kg)	4.0x10^{-5}	3800	WQP 2021

Table 5-6. Summary of Ambient Environmental Levels of HCH

Media	Low	High	Reference
β-HCH			
Outdoor air (ng/m^3)	0.00034	0.033	WQP 2021
Indoor air	No data		
Surface water (ng/L)	2.4x10^{-5}	80000	WQP 2021
Groundwater (ng/L)	0.94	2.5x10^6	WQP 2021
Drinking water	No data		
Food (ppm)		0.0027	Rogers et al. 1995
Soil and sediment (µg/kg)	4.0x10^{-5}	9390	WQP 2021
γ-HCH			
Outdoor air (ng/m^3)	<1.7	6.15	EPA 2021; Morgan et al. 2014
Indoor air (ng/m^3)	<0.09	18.5	Morgan et al. 2014
Surface water (ng/L)	0.04	100	Cole et al. 1984; Padma and Dickhut 2002
Groundwater (ng/L)	28	900	Adamski and Pugh 1996; Page 1981
Drinking water (ng/L)	0.01	319	Keith et al. 1976; Sandhu et al. 1978
Food (ppm)		0.0012	Rogers et al. 1995
Soil and sediment (µg/kg)	<0.02	150	Crockett et al. 1974; Sericano et al. 1990
δ-HCH			
Outdoor air	No data		
Indoor air	No data		
Surface water	No data		
Groundwater	No data		
Drinking water	No data		
Food (ppm)		0.0030	Rogers et al. 1995
Soil and sediment	No data		
HCH, technical grade			
Outdoor air	No data		
Indoor air	No data		
Surface water	No data		
Groundwater	No data		
Drinking water	No data		
Food (ppb)	No data		
Soil and sediment	No data		

Detections of HCH in air, water, and soil at NPL sites are summarized in Table 5-7.

Table 5-7. Hexachlorocyclohexanes Levels in Water, Soil, and Air of National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
α-HCH					
Water (ppb)	0.987	1.08	45.2	43	29
Soil (ppb)	3,390	2,970	156	52	28
Air (ppbv)	0.0016	0.0010	11	6	5
β-HCH					
Water (ppb)	0.68	1.09	18.7	34	22
Soil (ppb)	950	911	61.7	44	33
Air (ppbv)	0.0017	0.00080	7.1	2	2
γ-HCH					
Water (ppb)	0.63	0.963	40.6	63	37
Soil (ppb)	2,800	2,090	127	66	42
Air (ppbv)	0.0044	0.0038	29	7	7
δ-HCH					
Water (ppb)	0.57	1.18	34.5	29	19
Soil (ppb)	1,100	392	71.9	31	22
Air (ppbv)	0.0002	0.00016	1.5	3	2
HCH, technical grade					
Water (ppb)	0.86	3.0	24	6	5
Soil (ppb)	8,700	2,300	81	10	6
Air (ppbv)	0.000017	0.000017	1	2	1

[a]Concentrations found in ATSDR site documents from 1981 to 2019 for 1,867 NPL sites (ATSDR 2019). Maximum concentrations were abstracted for types of environmental media for which exposure is likely. Pathways do not necessarily involve exposure or levels of concern.

5.5.1 Air

HCH isomers have been detected in ambient air. The highest concentrations were found prior to γ-HCH agricultural use restrictions; available monitoring data after the restriction and ban of γ-HCH pesticides showed a gradual decrease and the results of the most recent monitoring studies are below the parts per billion range. The results of outdoor air monitoring studies are presented in Table 5-8. Precipitation samples, if available, are included in Table 5-8 because they reflect removal of atmospheric HCH.

One reference to indoor air monitoring was located in the literature search. In a study of preschool children's potential exposure to pesticides in North Carolina, indoor air samples from 13 daycare centers and 129 homes of the preschool children (ages 20–66 months) were collected between 2000 and 2001 (Morgan et al. 2014). γ-HCH was detected in ranges of below the limit of detection (<0.09 ng/m^3) to 18.5 ng/m^3 in the children's homes and <0.09–8.97 ng/m^3 in the preschools. Detection frequencies were 13 and 20%, respectively. Seventy-four percent of the homeowners reported applying insecticides at their homes, and 90% of these had applied an insecticide in the past year before sampling. Among the daycare centers, 62% reported using insecticides, and 88% of those reported usage within a year of sample collection (Morgan et al. 2014).

5.5.2 Water

Water monitoring data are presented in Table 5-9. HCH isomers have been detected in surface water, groundwater, and drinking water. The highest concentrations were found in groundwater below a facility that processed pesticides and stored wastes in unlined trenches until 1996 (Law et al. 2004). A study of the same site some years later still detected HCH isomers (Chartrand et al. 2015). Generally, surface water concentrations are lower than those detected in groundwater. Data from the EPA's Water Quality Portal (WQP), a system that maintains water monitoring data from stations across the United States, have been divided into two categories: prior to γ-HCH pesticide cancellation (years up to and including 2006) and post-cancellation (years after and including 2007) (WQP 2021). A decrease in surface and groundwater concentrations of α- and β-HCH can be seen in this dataset. A decrease of γ-HCH in surface water can be observed, possibly due to use limitations; trends for drinking water and groundwater are not as clear. Most recent monitoring data report concentrations below the parts per billion range for surface water and groundwater.

Table 5-8. Outdoor Air Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
α-HCH						
Alabama	Not reported	January–October 1996 and May 1997		0.092 ng/m^3		Jantunen et al. 2000
Eagle Harbor, Michigan; Sleeping Bear Dunes State Park, Michigan; Sturgeon Point, New York	Rural	1990–1997		0.110–0.140 ng/m^3		Cortes and Hites 2000
Chesapeake Bay	Rural and agricultural	April 2000–September 2003; excluding winter months	0.002–0.142 ng/m^3	0.026 ng/m^3	Gas phase; average of averages at three sites; detection frequency 99–100%	Goel et al. 2010
Chesapeake Bay	Rural and agricultural	April 2000 –September 2003; excluding winter months		0.0012 ng/m^3	Particulate phase; detection frequency 1%	Goel et al. 2010
Chesapeake Bay	Rural and agricultural	April 2000–September 2003; excluding winter months	0.2–11 ng/L	1.7 ng/L	Rainwater; average of averages at three sites; detection frequency 3–18%	Goel et al. 2010
Youngstown, Ohio	Urban/suburban	2000–2001		0.051 ng/m^3		Shen et al. 2004
Solomons, Maryland	Rural	2000–2001		0.091 ng/m^3		Shen et al. 2004
Wilmington, North Carolina	Urban/suburban	2000–2001		0.015 ng/m^3		Shen et al. 2004
Turkey Point, Florida	Rural	2000–2001		0.029 ng/m^3		Shen et al. 2004
Muscle Shoals	Suburban/rural	2000–2001		0.056 ng/m^3		Shen et al. 2004

Table 5-8. Outdoor Air Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
United States	Ambient air	2015–2018	0.00016–0.017 ng/m^3	0.0047 ng/m^3	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
β-HCH						
United States	Ambient air	2015–2018	0.00034–0.033 ng/m^3	0.004 ng/m^3	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
γ-HCH						
College Station, Texas	Rural	1979–1980	0.01–1.60 ng/m^3	0.23 ng/m^3	Ground level, ambient air	Atlas and Giam 1988
College Station, Texas	Rural	1979–1980	0.30–7.8 ng/L	2.81 ng/L	Rainwater samples	Atlas and Giam 1988
Adirondack Mountains, New York	Not reported	1985		0.509 ng/m^3	Troposphere samples	Knap and Binkley 1991
Newport News, Virginia	Not reported	1988		0.021 ng/m^3	Troposphere samples	Knap and Binkley 1991
Alabama	Not reported	January–October 1996 and May 1997		0.050 ng/m^3		Jantunen et al. 2000
Eagle Harbor, Michigan; Sleeping Bear Dunes State Park, Michigan; Sturgeon Point, New York	Rural	1990–1997		0.024–0.062 ng/m^3		Cortes and Hites 2000

Table 5-8. Outdoor Air Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
Lake Superior	Not reported	1984; wetfall season		3.0 ng/L	Rainwater samples, annual loading rate of 2.0 µg/m^2/year	Strachan 1988
Portland, Oregon	Urban	1982		0.45–1 ng/L	Rain and snow water	Pankow et al. 1984
Hawaii	Not reported	1970–1971	1–19 ng/L	5 ng/L	Rainwater	Bevenue et al. 1972
Youngstown, Ohio	Urban/suburban	2000–2001		0.049 ng/m^3		Shen et al. 2004
Solomons, Maryland	Rural	2000–2001		0.072 ng/m^3		Shen et al. 2004
Wilmington, North Carolina	Urban/suburban	2000–2001		0.026 ng/m^3		Shen et al. 2004
Turkey Point, Florida	Rural	2000–2001		0.031 ng/m^3		Shen et al. 2004
Muscle Shoals	Suburban/ agricultural	2000–2001		0.055 ng/m^3		Shen et al. 2004
North Carolina	Not reported	2000–2001	<0.09–0.11 ng/m^3		Air samples collected outside daycare centers; detection frequency 8%	Morgan et al. 2014
North Carolina	Not reported	2000–2001	<0.09–6.15 ng/m^3		Air samples collected outside students; homes; detection frequency 12%	Morgan et al. 2014
Chesapeake Bay	Rural and agricultural	April 2000–September 2003; excluding winter months	0.0012–0.382 ng/m^3	0.049 ng/m^3	Gas phase; average of averages at three sites; detection frequency 81–100%	Goel et al. 2010
Chesapeake Bay	Rural and agricultural	April 2000–September 2003; excluding winter months	0.0013–0.027 ng/m^3	0.024 ng/m^3	Particulate phase; average of averages at two sites; detection frequency 2–7%	Goel et al. 2010
Chesapeake Bay	Rural and agricultural	April 2000–September 2003; excluding winter months	0–35 ng/L	4.0 ng/L	Rainwater; average of averages at two sites; detection frequency 1–61%	Goel et al. 2010

Table 5-8. Outdoor Air Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
Texas	Various ambient air monitoring sites	January–December 2007		0.005 ng/m^3	Detected in 20 samples; below the limit of detection in 345 samples	EPA 2021
Texas	Various ambient air monitoring sites	January–December 2008		<1.7 ng/m^3	Below the limit of detection in 488 samples	EPA 2021
Texas	Various ambient air monitoring sites	January–June 2009		<1.7 ng/m^3	Below the limit of detection in 120 samples	EPA 2021

[a]Liquid unit conversion: 1 ng/L = 1 ppt = 0.001 ppb; gaseous unit conversion: ppbv = ([concentration ng/m^3] x 0.001) / 11.89, assuming standard temperature and pressure.

USGS = U.S. Geological Survey

Table 5-9. Water Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
α-HCH						
Lake Superior	Surface water	Spring 1997		2.8 ng/L		Marvin et al. 2004
Lake Erie	Surface water	Spring 1998		0.41 ng/L		Marvin et al. 2004
Lake Ontario	Surface water	Spring 1998		0.40 ng/L		Marvin et al. 2004
York River estuary	Surface water	June 1998–April 1999	~0.025–0.175 ng/L		Concentrations were lower in freshwater areas than areas with higher salinity	Padma and Dickhut 2002
United States	Surface water	1978–2006	$1.4x10^{-5}$–55,000 ng/L	150 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Surface water	2007–2021	$1.3x10^{-4}$–310 ng/L	5.4 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Groundwater	1981–2006	0.96–$2.5x10^{6}$ ng/L	7,000 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Groundwater	2007–2020	1.2–500 ng/L	270 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	58–$5.0x10^{5}$ ng/L	51,000 ng/L	Samples collected from 19 shallow wells; not detected (<20 ng/L) in 6 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	30–$4.2x10^{5}$ ng/L	44,000 ng/L	Samples collected from 15 deep wells; not detected (<20 ng/L) in 5 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004

Table 5-9. Water Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
Northeastern Florida	Surface water near active pesticide reformulating and packaging facility	2000	660–680 ng/L	670 ng/L	Three samples collected from creek adjacent to the site; not detected (<20 ng/L) in one sample; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
β-HCH						
United States	Surface water	1982–2006	$2.4x10^{-5}$–80,000 ng/L	340 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Surface water	2007–2018	$1.6x10^{-4}$–410 ng/L	9 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Groundwater	1981–2006	0.94–$2.5x10^{6}$ ng/L	7,200 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Groundwater	2007–2020	50–500 ng/L	290 ng/L	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	30–43,000 ng/L	12,000 ng/L	Samples collected from 19 shallow wells; not detected (<20 ng/L) in 8 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	82–820 ng/L	340 ng/L	Samples collected from 15 deep wells; not detected (<20 ng/L) in 8 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004

Table 5-9. Water Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
Northeastern Florida	Surface water near active pesticide reformulating and packaging facility	2000	38–440 ng/L	300 ng/L	Three samples collected from creek adjacent to the site	Law et al. 2004
γ-HCH						
New Jersey	Wells	Not reported (1981 or earlier)	NR–900 ng/L		1,076 wells, not detected in around half of samples	Page 1981
Chesterfield County, South Carolina	Rural drinking water	Not reported (1978 or earlier)	0–93 ng/L	23 ng/L		Sandhu et al. 1978
Hampton, South Carolina	Rural drinking water	Not reported (1978 or earlier)	0–319 ng/L	147 ng/L		Sandhu et al. 1978
Cincinnati, Ohio	Drinking water	Not reported (1976 or earlier)		0.01 ng/L		Keith et al. 1976
Oahu, Hawaii	Drinking water	1970–1971		0.2 ng/L		Bevenue et al. 1972
Ozark Plateaus Province of Arkansas, Kansas, Missouri, and Oklahoma	Groundwater	April–September 1993	28 and 32 ng/L		Detected in two samples from domestic wells	Adamski and Pugh 1996
Connecticut	Drinking water well	Not reported (1999 or earlier)		60 ng/L		Eitzer and Chevalier 1999

Table 5-9. Water Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
United States	Drinking water	1998–2005	1–690 ng/L	62 ng/L	Samples collected from 44 states across the United States; ranges and averages do not include samples reported to be below the method reporting level.	EPA 2010
Washington, D.C. and Denver, Colorado	Surface water	Not reported (1984 or earlier)	52–100 ng/L			Cole et al. 1984
Niagara River	Surface water	1980–1981		2.1 ng/L	Mean of 99% samples	Kuntz and Warry 1983
Lake Michigan tributary streams	Surface water	Not reported (1974 or earlier)	ND–150 ng/L			EPA 1974
United States	Surface water	Not reported (1985 or earlier)		Median: 20 ng/L	Detected in 27% of 4,505 samples	Staples et al. 1985
Lake Ontario	Surface water	1983	0.806–1.85 ng/L			Biberhofer and Stevens 1987
Patuxent River	Surface water	1995	1.0 ng/L			Harman-Fetcho et al. 1999
Lake Superior	Surface water	Spring 1997		0.38 ng/L		Marvin et al. 2004
Lake Erie	Surface water	Spring 1998		0.32 ng/L		Marvin et al. 2004
Lake Ontario	Surface water	Spring 1998		0.24 ng/L		Marvin et al. 2004
York River estuary	Surface water	June 1998–April 1999	~0.04–0.21 ng/L		Concentrations were higher in freshwater areas than areas with higher salinity	Padma and Dickhut 2002
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	110–6.6×10^5 ng/L	99,000 ng/L	Samples collected from 19 shallow wells; not detected (<20 ng/L) in 11 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004

Table 5-9. Water Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	120–$3.6x10^5$ ng/L	73,000 ng/L	Samples collected from 15 deep wells; not detected (<20 ng/L) in 10 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
Northeastern Florida	Surface water near active pesticide reformulating and packaging facility	2000	440–470 ng/L	460 ng/L	Three samples collected from creek adjacent to the site; not detected (<20 ng/L) in one sample; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
δ-HCH						
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	40–$5.7x10^5$ ng/L	74,000 ng/L	Samples collected from 19 shallow wells; not detected (<20 ng/L) in 8 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
Northeastern Florida	Groundwater below active pesticide reformulating and packaging facility	2000	36–$2.9x10^5$ ng/L	26,000 ng/L	Samples collected from 15 deep wells; not detected (<20 ng/L) in 3 wells; mean and ranges do not reflect samples reported as not detected/below detection limit	Law et al. 2004
Northeastern Florida	Surface water near active pesticide reformulating and packaging facility	2000	55–970 ng/L	640 ng/L	Three samples collected from creek adjacent to the site	Law et al. 2004
HCH, mixture						
Northeastern Florida	Groundwater below contaminated site	Not reported (2015 or earlier)	30,000–$4.2x10^5$ ng/L		Range reported for α-, γ-, and δ-HCH	Chartrand et al. 2015

[a]Liquid unit conversion: 1 ng/L = 1 ppt = 0.001 ppb.

USGS = U.S. Geological Survey

5.5.3 Sediment and Soil

HCH has been detected in soil and sediment as a result of agricultural use. Soil and sediment monitoring data are presented in Table 5-10. Data from the EPA's WQP has been divided into two categories to reflect potential detection decreases as a result of the γ-HCH pesticide cancellation in 2006 (WQP 2021). A clear trend could not be discerned from the data, possibly due to the large ranges reflecting differences in land use. Most recent monitoring data report concentrations at the parts per billion range.

5.5.4 Other Media

HCH isomers have been detected in aquatic organisms; the results of these monitoring studies are summarized in Table 5-11. Data on terrestrial organism monitoring were not located. Schmitt et al. (1985) reported the results of a monitoring study of fish tissues from 107 freshwater stations in the United States from 1976 to 1981, which supported a decline in tissue occurrence of detectable α- and γ-HCH residues in aquatic organisms. α- and β-HCH have been detected in organisms as recently as 2018, however (WQP 2021). The most recent monitoring studies have detected HCH isomers in aquatic organisms in the parts per billion range.

Historically, as a result of pesticide use, γ-HCH was detected in meat, vegetables, and other food items, both imported to and produced in the United States. Due to the discontinued agricultural use of γ-HCH by the United States and many other countries, residues are typically no longer detected in food products. γ-HCH was detected in 5 out of 612 imported rice samples at a maximum concentration of 0.03 ppm during an FDA pesticide monitoring study conducted in 1993–1994 (Roy et al. 1997). A 10-year (1982–1991) FDA study of ready-to-eat foods commonly consumed in the United States showed that α-, β-, δ-, and γ-HCH were frequently detected (Rogers et al. 1995). The results of this study reported average concentrations of 0.0010, 0.0027, 0.0030, and 0.0012 ppm for α-, β-, δ-, and γ-HCH isomers, respectively, in 243 ready-to-eat foods. HCH isomers were also detected in the following feed types formulated for infants and toddlers and in adult diet foodstuffs: whole milk and other dairy products; meat, fish, and poultry; oils and fats; vegetables; and sugars and adjuncts (Gartrell et al. 1986a, 1986b).

Table 5-10. Soil and Sediment Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
α-HCH						
Alabama	Soil	Not reported (1999 or earlier)	0–0.269 µg/kg		Detected in 26 of 39 soils from 6 regions	Harner et al. 1999
Sequoia National Park, Rocky Mountain National Park, Mt. Rainier National Park, Denali National Park, Noatak National Preserve, and Gates of the Arctic National Park and Preserve	Sediment core from deepest point in several lakes	2003–2005	<0.8 µg/kg		Not detected	Genualdi et al. 2011
United States	Soil and sediment	1982–2006	4.0×10^{-5}–3,800 µg/kg	5.7 µg/kg	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Soil and sediment	2007–2020	1.2×10^{-2}–1,060 µg/kg	2.7 µg/kg	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
β-HCH						
United States	Soil and sediment	1989–2006	4.0×10^{-5}–4,230 µg/kg	8.8 µg/kg	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021

Table 5-10. Soil and Sediment Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
United States	Soil and sediment	2007–2020	$1.3x10^{-2}$–9,390 µg/kg	11 µg/kg	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
γ-HCH						
Alabama, Arkansas, Georgia, Illinois, Iowa	Soil	Not reported (1974 or earlier)	10–150 µg/kg	52 µg/kg		Crockett et al. 1974
Alabama	Soil	Not reported (1999 or earlier)	0–1.07 µg/kg		Detected in 26 of 39 soils from 6 regions	Harner et al. 1999
Niagara River	Suspended sediment	Not reported (1983 or earlier)		2 µg/kg	Detection frequency 33%	Kuntz and Warry 1983
Lake Ontario	Settling particulates	1982		2.4 µg/kg		Oliver and Charlton 1984
James River, Virginia	Creek sediments	1976	7.3–8.5 µg/kg			Saleh et al. 1978
Gulf of Mexico	Sediment	1987	<0.02–1.74 µg/kg	0.07 µg/kg	Detection frequency 19%	Sericano et al. 1990
Around the Great Lakes	Sediment	May 1989	<0.10–0.99 µg/kg wet weight			Verbrugge et al. 1991
Indian River Lagoon, Florida	Sediment from impoundments along the river	Not reported (1992 or earlier)	9.4–34.4 µg/kg		33 sediment samples from 11 impoundment	Wang et al. 1992

Table 5-10. Soil and Sediment Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Geographic type	Date(s)	Range[a]	Mean[a]	Notes	Reference
HCH, mixture						
South Carolina	0–10 cm surface soils from cotton fields	November 1999	0.1–0.54 µg/kg dry weight	0.27 µg/kg dry weight	Reported as sum of α-, β-, and γ-isomers; not detected (<0.1 µg/kg dry weight) in 10 of 16 samples; mean and ranges do not reflect samples reported as not detected/below detection limit	Kannan et al. 2003
Georgia	0–10 cm surface soils from cotton fields	December 1999	0.16–0.49 µg/kg dry weight	0.33 µg/kg dry weight	Reported as sum of α-, β-, and γ-isomers; not detected (<0.1 µg/kg dry weight) in 14 of 16 samples; mean and ranges do not reflect samples reported as not detected/below detection limit	Kannan et al. 2003
United States	Sediment	2008	2.1–11 µg/kg	6.03 µg/kg	HCH isomer or mixture not specified; data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
United States	Sediment	2017	0.47–1.26 µg/kg	0.75 µg/kg	HCH isomer or mixture not specified; data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021

[a]Solid unit conversion: 1 µg/kg = 1 ppb.

USGS = U.S. Geological Survey

Table 5-11. Organism Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Species	Date(s)	Range	Mean	Notes	Reference
α-HCH						
Southwestern and Midwestern United States	Freshwater Fish	1980–1981	0.03–0.04 ng/g		Highest concentrations detected from 107 monitoring stations across the United States	Schmitt et al. 1985
United States	Freshwater fish	1984	NR–10 ng/g	<10 ng/g		Schmitt et al. 1990
Louisiana section of the Mississippi River	Blue crab, cobia, flathead catfish, freshwater drum, long-nose gar, red drum, red snapper, river shrimp, small-mouth buffalo, spotted gar	1990–1994		Not detected		Watanabe et al. 2003
Louisiana section of the Mississippi River	Bigmouth buffalo	1990–1994		2.4 ng/g	Average detection in 1 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Blue catfish	1990–1994	0.333–26.3 ng/g		Range of average detections in 3 of 4 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Carp	1990–1994		31.1 ng/g	Range of average detections in 3 of 4 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Channel catfish	1990–1994	1.83–7.23 ng/g		Range of average detections in 2 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Crawfish	1990–1994		4.25 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Largemouth bass	1990–1994		1.00 ng/g	Average detection in 1 of 2 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Striped bass	1990–1994		2.88 ng/g	Average detection in 1 of 2 sampling years; not detected in other years	Watanabe et al. 2003

Table 5-11. Organism Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Species	Date(s)	Range	Mean	Notes	Reference
Louisiana section of the Mississippi River	White bass	1990–1994		1.44 ng/g	Average detection in 1 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	White crappie	1990–1994		1.75 ng/g	Average detection in 1 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Southwestern Michigan	Adult green frogs	1998		0.02 ng/g	0.04 ppb detected in juvenile frogs	Gilliland et al. 2001
Gulf of California	Clams (*Chione californiensis*)	Not reported (2015 or earlier)	<0.005–1.77 ng/g wet weight		Detection frequency 16.7% in 1 of 3 study areas, not detected in other areas; adipose tissue samples; surrounding area has primarily agricultural activity	Vargas-Gonzalez et al. 2016
Lake Apopka, Florida	Largemouth bass (*Micropterus salmoides*)	March 2013		Not detected	Limit of quantification 0.1–0.5 ng/g wet weight	Dang et al. 2016
United States	Freshwater fish	2018	2–150 ng/g wet weight	17 ng/g wet weight	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
β-HCH						
Upper Steele Bayou, Mississippi	Fish	1988	ND–20 ng/g wet weight			Ford and Hill 1991
Upper Steele Bayou, Mississippi	Snakes	1988	ND			Ford and Hill 1991
Southwestern Michigan	Adult green frogs	1998		0.01 ng/g	Not detected in juvenile frogs	Gilliland et al. 2001
Louisiana section of the Mississippi River	Cobia, long-nose gar, red drum, red snapper	1990–1994		Not detected		Watanabe et al. 2003
Louisiana section of the Mississippi River	Bigmouth buffalo	1990–1994	2.25–11.2 ng/g		Range of average detection in 3 of 3 sampling years	Watanabe et al. 2003

Table 5-11. Organism Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Species	Date(s)	Range	Mean	Notes	Reference
Louisiana section of the Mississippi River	Blue catfish	1990–1994	3.67–8.33 ng/g		Range of average detection in 3 of 4 sampling years; not detected in other year	Watanabe et al. 2003
Louisiana section of the Mississippi River	Blue crab	1990–1994	4.00–11.0 ng/g		Range of average detections in 2 of 2 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Carp	1990–1994	5.00–11.3 ng/g		Range of average detections in 2 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Channel catfish	1990–1994	0.333–7.77 ng/g		Range of average detections in 3 of 4 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Crawfish	1990–1994		27.5 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Flathead catfish	1990–1994		0.500 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Freshwater drum	1990–1994		0.333 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Largemouth bass	1990–1994		2.00 ng/g	Average detection in 1 of 2 sampling years; not detected in other year	Watanabe et al. 2003
Louisiana section of the Mississippi River	River shrimp	1990–1994		8.67 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Small-mouth buffalo	1990–1994	0.25–2.00 ng/g		Range of average detection in 3 of 4 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Spotted gar	1990–1994		39.0 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Striped bass	1990–1994	5.50–57.9 ng/g		Range of average detection in 2 of 2 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	White bass	1990–1994		3.11 ng/g	Average detection in 1 of 3 sampling years; not detected in other years	Watanabe et al. 2003

Table 5-11. Organism Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Species	Date(s)	Range	Mean	Notes	Reference
Louisiana section of the Mississippi River	White crappie	1990–1994	1.00–11.0 ng/g		Range of average detection in 2 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Gulf of California	Clams (*C. californiensis*)	Not reported (2015 or earlier)		Not detected	Limit of detection 0.01 ng/g wet weight Adipose tissue samples; surrounding area has primarily agricultural activity	Vargas-Gonzalez et al. 2016
Lake Apopka, Florida	Largemouth bass (*M. salmoides*)	March 2013		Not detected	Limit of quantification 0.1–0.5 ng/g wet weight	Dang et al. 2016
United States	Freshwater fish	2018	1–53 ng/g wet weight	10 ng/g wet weight	Data collected by USGS monitoring stations across the United States; mean and ranges do not reflect samples reported as not detected/below detection limit	WQP 2021
γ-HCH						
Gulf of Mexico	Oyster	1987	<0.25–9.06 ng/g	1.74 ng/g	Detection frequency 80%	Sericano et al. 1990
United States	Freshwater fish	1980–1981	0.02–0.03 ng/g		Whole body concentrations were >0.01 ng/g at 1 of 107 monitoring stations	Schmitt et al. 1985
United States	Freshwater fish	1984	NR–40 ng/g	<10 ng/g		Schmitt et al. 1990
Southwestern Michigan	Adult green frogs	1998		0.07 ng/g	Not detected in juvenile frogs	Gilliland et al. 2001
Louisiana section of the Mississippi River	Blue crab, channel catfish, cobia, crawfish, flathead catfish, freshwater drum, long-nose gar, red drum,	1990–1994		Not detected		Watanabe et al. 2003
Louisiana section of the Mississippi River	Bigmouth buffalo	1990–1994	1.00–1.80 ng/g		Range of average detection in 2 of 3 sampling years not detected in other year	Watanabe et al. 2003

Table 5-11. Organism Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Species	Date(s)	Range	Mean	Notes	Reference
Louisiana section of the Mississippi River	Blue catfish	1990–1994	2.25–26.5 ng/g		Range of average detection in 3 of 4 sampling years; not detected in other year	Watanabe et al. 2003
Louisiana section of the Mississippi River	Carp	1990–1994	0.714–5.00 ng/g		Range of average detections in 2 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Largemouth bass	1990–1994		1.00 ng/g	Average detection in 1 of 2 sampling years; not detected in other year	Watanabe et al. 2003
Louisiana section of the Mississippi River	Red snapper	1990–1994		0.333 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	River shrimp	1990–1994		1.67 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Small-mouth buffalo	1990–1994	0.250–7.00 ng/g		Range of average detection in 4 of 4 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Spotted gar	1990–1994		857 ng/g	Average detection in 1 of 1 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	Striped bass	1990–1994	0.500–1.25 ng/g		Range of average detection in 2 of 2 sampling years	Watanabe et al. 2003
Louisiana section of the Mississippi River	White bass	1990–1994		7.56 ng/g	Average detection in 1 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Louisiana section of the Mississippi River	White crappie	1990–1994	0.750–1.20 ng/g		Range of average detection in 2 of 3 sampling years; not detected in other years	Watanabe et al. 2003
Gulf of California	Clams (*C. californiensis*)	Not reported (2015 or earlier)		Not detected	Limit of detection 0.005 ng/g wet weight; adipose tissue samples; surrounding area has primarily agricultural activity	Vargas-Gonzalez et al. 2016

Table 5-11. Organism Monitoring Data for Hexachlorocyclohexanes (HCHs)

Locations	Species	Date(s)	Range	Mean	Notes	Reference
Lake Apopka, Florida	Largemouth bass (*M. salmoides*)	March 2013		0.8 ng/g wet weight (gastrointestinal tract) 1.6 ng/g wet weight (liver) 2.6 ng/g wet weight (kidney) 0.8 ng/g wet weight (spleen) 4.8 ng/g wet weight (brain) 1.8 ng/g wet weight (gonad) 1.3 ng/g wet weight (muscle)	117.8 ng/g lipid (gastrointestinal tract) 29.4 ng/g lipid (liver) 742.6 ng/g lipid (kidney) 102.0 ng/g lipid (spleen) 144.1 ng/g lipid (brain) 77.6 ng/g lipid (gonad) 125.2 ng/g lipid (muscle)	Dang et al. 2016
δ-HCH						
Southwestern Michigan	Adult green frogs	1998		0.03 ng/g	Not detected in juvenile frogs	Gilliland et al. 2001
Gulf of California	Clams (*C. californiensis*)	Not reported (2015 or earlier)	<0.01–1.97 ng/g wet weight		Detection frequency 16.7% in 1 of 3 study areas, not detected in other areas; adipose tissue samples; surrounding area has primarily agricultural activity	Vargas-Gonzalez et al. 2016
Lake Apopka, Florida	Largemouth bass (*M. salmoides*)	March 2013		Not detected	Limit of quantification 0.1–0.5 ng/g wet weight	Dang et al. 2016

[a]Organism concentration unit conversion: 1 ng/g = 1 ppb.

NR = not reported; USGS = U.S. Geological Survey

γ-HCH residues were detected in fat samples of domestic farm animals collected in Ontario, Canada, in 1986–1988. Mean concentrations in fat from chickens, turkeys, beef, lamb, and pork ranged from 0.012 to 0.032 ppm; the mean concentration in hen eggs was 0.008 ppm (Frank et al. 1990). A pesticide residue screening program carried out by the H.E.B. Food Stores of San Antonio between 1989 and 1991 detected γ-HCH in 4 of 429 onion samples (detection limit 0.02 ppm); however, none of the positive samples exceeded the action level for this commodity (Schattenberg and Hsu 1992). γ-HCH was detected at levels of ≤10 ppm in 6 out of 5,784 fruit and vegetable commodities analyzed in Canada from 1992 to 1994 (Neidert and Saschenbrecker 1996). α-, β-, and γ-HCH were detected in butter samples from the United States at mean levels of 0.38, 0.42, and 0.78 ppb, respectively (Kalantzi et al. 2001). HCH isomers were also detected in butter samples from 20 other countries, with the highest levels being observed in a single butter sample from India with reported concentrations of 98, 108, and 164 ppb for α-, β-, and γ-HCH, respectively (Kalantzi et al. 2001).

Based on the most recent pesticide residue monitoring results published by the FDA from 2018, no γ-HCH residues were detected on food products produced in or imported to the United States (FDA 2020a). The products sampled were broadly encompassing of domestic and imported food and agricultural commodities, and included fruits, vegetables, grains, beans, nuts, honey, milk, and meat, amongst many other categories. A recent study, however, detected averages of 0.22 ppb α-HCH and 0.77 ppb γ-HCH in tobacco products (n=20; cigarettes from one pack were pooled for analysis) purchased in the United States (Quadroni and Bettinetti 2019). It is unclear if these products were domestic or imported.

Strategies exist to reduce pesticide residues on food products. γ-HCH residues on tomatoes decreased by 23.9% 15 days after application of the pesticide (from 0.1956 to 0.1488 ppm). Processing the tomatoes (e.g., pureeing, making tomato juice) reduced the residue levels by 100% after the waiting period; however, washing the tomatoes reduced the residues by up to 55.9% (Bessar et al. 1991). An analysis of pesticide residues in green coffee and after roasting indicated that technical-grade HCH was found in green coffee at concentrations ranging from <0.005 to 0.204 ppm. However, storage and roasting reduced the pesticide residues by 60–67% and up to 98%, respectively, with darker roasting resulting in the greatest reduction (McCarthy et al. 1992).

5.6 GENERAL POPULATION EXPOSURE

Exposure of the general population to HCH has declined steadily since its use as a pesticide was discontinued. Human exposure to γ-HCH may result from environmental exposures to contaminated water or soil, or possible ingestion of small amounts in drinking water. Historically, γ-HCH and its most persistent metabolite, β-HCH, have been detected in blood, adipose tissue, and breast milk. Based on the most recent U.S. population survey, γ-HCH is generally no longer detected in blood, but β-HCH still is (CDC 2019). This is consistent with the decreased general population exposure to γ-HCH as a pesticide. Medicinal exposure to γ-HCH can occur from prescription scabies and lice treatments. An analysis of data from 238 families in Missouri between June 1989 and March 1990 indicated that 9.2% of the families reported using Kwell shampoo (contains γ-HCH) for lice control on children (Davis et al. 1992). In general, accidental or intentional ingestion of these products would lead to the highest exposures. Worker exposure constitutes the next highest exposure population, although worker exposure is decreasing in both the number of workers exposed and the levels of exposure. Lastly, the general population receives the lowest levels, which occur mainly from ingestion of foods and water with γ-HCH residues. Living near a waste disposal site contaminated with γ-HCH will also increase the likelihood of exposure.

Ingestion of food containing pesticide residue, historically a significant route of exposure, is no longer expected to be a likely route of non-medicinal human exposure to γ-HCH. During studies conducted between 1982 and 1991, γ-HCH was detected in 4–6% of the foods collected in eight market basket surveys from different regions of the United States (Gunderson 1988, 1995a, 1995b). The most recent results of this survey reported no detections (limit of detection 0.4–2.8 ppb) in foods surveyed in 2017 for α-, β-, γ-, or δ-HCH (FDA 2020b). γ-HCH was also not detected in domestic or imported food products in the United States by the FDA (FDA 2020a). Foods representative of consumption patterns by eight infant and adult population groups were prepared for consumption prior to analysis in a revision to FDA's Total Diet Studies methodology. The estimated mean daily intakes (ng/kg body weight/day) of α-, β-, and γ-HCH for these groups continuously decreased between all study periods (1982–1984, 1984–1986, and 1986–1991). From 1986 to 1991 daily intakes ranged from 0.5 to 2.7 ng/kg/day for α-HCH, were all <1 ng/kg/day for β-HCH, and ranged from 0.6 to 3.2 ng/kg/day for γ-HCH (Gunderson 1988, 1995a, 1995b). An estimated γ-HCH daily dietary intake based on 2003 FDA pesticide residue monitoring data for fruits and vegetables was determined to be trace only for domestic produce and 0.00754 ng/kg/day for imported produce (Katz and Winter 2009). Because γ-HCH and its isomers were not detected in most recently available food monitoring data, current daily intake can be assumed to be negligible.

A small degree of exposure to γ-HCH from drinking water may be possible. α- and β-HCH have been detected in recent surface water and groundwater samples, from 2007 to 2020, although these samples may not represent drinking water sources (WQP 2021). γ-HCH was detected in drinking water samples collected for the 1998 to 2005 EPA 6-year review of drinking water quality (EPA 2010). Data for HCH were not reported for the most recent available period, 2006–2011.

Contaminated soils, which have been sampled as recently as 2020, may present another exposure pathway (WQP 2021). Studies in which soils containing 10 ppm radiolabeled γ-HCH were added to human skin samples at quantities that exceeded monolayer coverage (5 mg soil/cm^2 skin) demonstrated mean γ-HCH absorptions of 1.04% from sandy soils and 1.64% from silt soils (Duff and Kissel 1996). However, data from soil absorption studies can vary due to factors such as the amount of soil added to skin, exposure time, and possible evaporation of the contaminant.

The results of biomonitoring studies can be used as indicators of human exposures to HCH. The National Human Adipose Tissue Survey (NHATS) conducted in 1982 showed that β-HCH (the most prevalent HCH isomer in fatty tissue) was detected in 87% of 46 composite samples at concentrations <19–570 ng/g (ppb) (EPA 1986). It was detected most often in postmortem samples collected from individuals from the southern United States. In another survey conducted in 1970–1975, β-HCH was detected in >90% of the postmortem human adipose tissue samples at an average level of 300 ppb (Kutz et al. 1979). In a review of the NHATS data available from 1970 to 1983, EPA (1985c) reported that the estimated 1983 national median level of β-HCH was 80 ppb, in comparison to the historic level of 140 ppb. The median level had decreased over time, but the compound continued to be detected in nearly 100% of the population surveyed. Median levels were highest in the South census region and tended to increase with age but had not been found to differ across the sexes or racial groups. A further analysis of the NHATS data indicated that average β-HCH concentrations in fat had decreased from 0.45 ppm in 1970 to approximately 0.16 ppm since 1981 (Kutz et al. 1991). In a similar study in Japan, levels of HCHs in the adipose tissue of Japanese males increased from the late 1940s to 1966, coinciding with an increased annual production of HCH, and began dropping when HCHs were banned in 1971, with only the only the more persistent β-HCH isomer detected after 1974 (Loganathan et al. 1993). Recent adipose tissue concentrations in the United States were not located, but the trend towards lower concentrations may have continued following the discontinued use of γ-HCH as a pesticide.

A comparison of the levels of α- and β-HCH in the whole blood and biopsy fat of 25 patients showed median levels of <0.04 ng/g (maximum, <0.04 ng/g [LOD]) and 0.13 ng/g (maximum, 2.60 ng/g) for the blood and 1.1 ng/g (maximum, 9.6 ng/g) and 18.0 ng/g (maximum, 748.6 ng/g) for the fat tissue, respectively (Mes 1992). A further comparison of β-HCH levels in breast milk and adipose tissue samples was made for populations living near the Great Lakes (Canada only) and in other Canadian regions. Mean β-HCH levels in breast milk (0.6 ng/g, n=70 samples) and adipose tissue (23.4 ng/g, n=16 samples) were lower near the Great Lakes than in other parts of Canada (0.8 [n=305 samples] and 30.8 ng/g [n=90 samples], respectively) (Mes and Malcolm 1992). In addition, studies indicate that γ-HCH can also be present in breast milk at a previously reported average level of 0.006 ppm in Alberta, Canada (Currie et al. 1979). In a study of 50 donors of breast milk in Oahu, Hawaii, Takahashi et al. (1981) demonstrated HCH in 82% of the samples at a mean level of 81 ppb within a range of 0–480 ppb, expressed in terms of extractable lipid.

γ-HCH was one of the most frequently detected pesticides in the blood of Virginia residents, although the number of individuals sampled was not identified (Griffith and Blanke 1975). γ-HCH blood concentrations were the highest in residents of the middle age group (41–60 years). Some of the frequency of γ-HCH occurrence in the state was attributed to its common use in commercial vaporizers and its presence in cigarette smoke (Griffith and Blanke 1975). NHANES analyzed blood and urine specimens for the presence of HCH isomers. β-HCH was detected in approximately 13.9% of the U.S. population (12–74 years) in the Northeast, Midwest, and South. The median level for the 91% quantifiable positive results was 1.7 ppb (Murphy and Harvey 1985).

In a more recent study (1999–2000) of pesticide serum concentrations in pregnant Latina women living in an agricultural community in California, median serum levels were 36.9 ng/g lipid for β-HCH and 1.1 ng/g lipid for γ-HCH (Bradman et al. 2007). The median serum concentration of β-HCH was 5.9 ng/g lipid in 48 mothers enrolled in the California Childhood Leukemia Study in 2006–2007 (Whitehead et al. 2015). Maternal and blood cord samples were collected predominantly from Latina women, who were in their second or third trimester of pregnancy, as part of a study from October 2010 to June 2011 in San Francisco, California. Sixty-seven percent of maternal blood samples were above the method detection limit for β-HCH (5 ng/L wet weight). The lipid adjusted median cord:maternal serum ratio was 1.0, suggesting equivalent exposures for the fetus and the mother. (Morello-Frosch et al. 2016). In another study of 10 whole blood samples obtained from a blood donation center in Palo Alto, California, α-HCH was detected in all samples, β- and δ-HCH were detected in 60% of samples, and γ-HCH was detected in

80% of samples. Whole blood concentrations were 0.291–0.828, 0.544–1.15, 0.250–0.694, and 0.550–1.26 ng/g for α-, β-, γ-, and δ-HCH, respectively (Hao et al. 2020).

The Centers for Disease Control and Prevention (CDC) completed its Fourth National Report on Human Exposure to Environmental Chemicals that was derived from data obtained from NHANES (CDC 2019). The first report on 27 chemicals was issued in March 2001. This fourth report, released in January 2019, presents blood and urine levels of environmental chemicals from a sample of people who represent the noninstitutionalized, civilian U.S. population during 2-year study periods over 1999–2014. Lipid serum levels of β- and γ-HCH from the most recent available study period, 2013–2014 and 2011–2012, respectively, are summarized in Table 5-12. Serum monitoring was not conducted in the following NHANES study periods.

Table 5-12. Geometric Mean of the Serum Concentration (ng/g) of β-Hexachlorocyclohexane (β-HCH) (2013–2014) and γ-Hexachlorocyclohexane (γ-HCH) (2011–2012) in the U.S. Population

Population group (sex, age)	Geometric mean	Unadjusted standard error	Sample size (pools)[a]
β-HCH			
Non-Hispanic white			
Male, 12–19 years	NA[b]	NA	8
Male, 20–39 years	NA	NA	14
Male, 40–59 years	NA	NA	14
Male, ≥60 years	3.67	1.05	16
Female, 12–19 years	NA	NA	6
Female, 20–39 years	NA	NA	15
Female, 40–59 years	3.29	0.45	14
Female, ≥60 years	18.4	5.2	20
Non-Hispanic black			
Male, 12–19 years	NA	NA	6
Male, 20–39 years	NA	NA	7
Male, 40–59 years	3.70	1.11	7
Male, ≥60 years	6.90	1.67	9
Female, 12–19 years	NA	NA	6
Female, 20–39 years	NA	NA	8
Female, 40–59 years	5.61[c]	1.73	8
Female, ≥60 years	32.9	8.1	8
Mexican American			
Male, 12–19 years	NA	NA	7
Male, 20–39 years	5.52[c]	3.05	5
Male, 40-59 years	4.54	1.08	5

Table 5-12. Geometric Mean of the Serum Concentration (ng/g) of β-Hexachlorocyclohexane (β-HCH) (2013–2014) and γ-Hexachlorocyclohexane (γ-HCH) (2011–2012) in the U.S. Population

Population group (sex, age)	Geometric mean	Unadjusted standard error	Sample size (pools)[a]
Male, ≥60 years	8.64	1.27	4
Female, 12–19 years	1.35	0.25	7
Female, 20–39 years	8.28	1.76	5
Female, 40–59 years	11.4	1.1	6
Female, ≥60 years	44.4	9.6	5
All Hispanic			
Male, 12–19 years	NA	NA	10
Male, 20–39 years	4.38[c]	2.08	8
Male, 40–59 years	4.35	0.98	9
Male, ≥60 years	10.7	1.4	7
Female, 12–19 years	NA	NA	10
Female, 20–39 years	5.58	1.46	9
Female, 40–59 years	10.3	1.5	10
Female, ≥60 years	37.8	6.9	9
Asian			
Male, 12–19 years	23.0[c]	17.7	3
Male, 20–39 years	25.7	0.7	4
Male, 40–59 years	143	22	4
Male, ≥60 years	57.4[c]	41	3
Female, 12–19 years	9.01[c]	5.74	3
Female, 20–39 years	44.7[c]	14.1	5
Female, 40–59 years	227[c]	115	7
Female, ≥60 years	242[c]	101	3
γ-HCH			
Non-Hispanic white			
Male, 12–19 years	NA	NA	6
Male, 20–39 years	NA	NA	12
Male, 40–59 years	NA	NA	12
Male, ≥60 years	NA	NA	12
Female, 12–19 years	NA	NA	5
Female, 20–39 years	NA	NA	13
Female, 40–59 years	NA	NA	11
Female, ≥60 years	NA	NA	14
Non-Hispanic black			
Male, 12–19 years	NA	NA	7
Male, 20–39 years	NA	NA	9
Male, 40–59 years	NA	NA	7
Male, ≥60 years	NA	NA	9
Female, 12–19 years	NA	NA	6

Table 5-12. Geometric Mean of the Serum Concentration (ng/g) of β-Hexachlorocyclohexane (β-HCH) (2013–2014) and γ-Hexachlorocyclohexane (γ-HCH) (2011–2012) in the U.S. Population

Population group (sex, age)	Geometric mean	Unadjusted standard error	Sample size (pools)[a]
Female, 20–39 years	NA	NA	8
Female, 40–59 years	NA	NA	8
Female, ≥60 years	NA	NA	8
Mexican American			
Male, 12–19 years	NA	NA	5
Male, 20–39 years	NA	NA	4
Male, 40–59 years	NA	NA	4
Male, ≥60 years	NA	NA	2
Female, 12–19 years	NA	NA	4
Female, 20–39 years	NA	NA	4
Female, 40–59 years	NA	NA	3
Female, ≥60 years	NA	NA	3
All Hispanic			
Male, 12–19 years	NA	NA	7
Male, 20–39 years	NA	NA	8
Male, 40–59 years	NA	NA	7
Male, ≥60 years	NA	NA	6
Female, 12–19 years	NA	NA	7
Female, 20–39 years	NA	NA	8
Female, 40–59 years	NA	NA	7
Female, ≥60 years	NA	NA	7
Asian			
Male, 12–19 years	NA	NA	3
Male, 20–39 years	NA	NA	6
Male, 40–59 years	NA	NA	6
Male, ≥60 years	NA	NA	4
Female, 12–19 years	NA	NA	4
Female, 20–39 years	NA	NA	6
Female, 40–59 years	NA	NA	6
Female, ≥60 years	NA	NA	3

[a]Each pool contained serum from eight people.
[b]NA = not available; proportion of results below limit of detection (1.3 ng/g lipid for β-HCH and 0.92 ng/g lipid for γ-HCH).was too high to provide a valid result.
[c]Standard error of the mean is >30%.

[a]Source: CDC 2019

Factors such as age, dietary habits, and residence can influence the body burden of γ-HCH in exposed individuals. In one study, it was shown that women between the ages of 26 and 34 years who lived in a

rural area of India and were nonvegetarians tended to show higher body levels of γ-HCH than other Indian women who lived in an urban area or who were vegetarians (Saxena et al. 1981a). The higher levels of γ-HCH in women at an older child-bearing age suggest that a longer life span may cause a greater accumulation of pesticide in the body. Higher pesticide levels were found in mutton, eggs, and chicken, which are common in nonvegetarian meals; therefore, there tended to be a higher level of γ-HCH in the bodies of nonvegetarians. In another study, when corrected for age and BMI, vegans had an almost statistically significantly lower (p=0.076) mean β-HCH plasma concentration, not adjusted for lipids, than omnivores. Mean β-HCH plasma concentrations were 6.151 ng/g lipid for vegans and 5.720 ng/g lipid for omnivores (Arguin et al. 2010). In a study of hair samples, 460 ng/g γ-HCH was detected in samples from people who worked as pesticide applicators and 40 ng/g γ-HCH was detected in samples from people who lived close to farms in Atlanta, Georgia. Hair collected from people in Houston, Texas, representing urban environmental exposure, had 1,500 ng/g γ-HCH detected (Smith-Baker and Saleh 2011). The study authors did not suggest an explanation for the higher levels in the samples from environmentally exposed persons in Houston, Texas compared with levels in pesticide applicators in Atlanta, Georgia; however, the sample sizes were very small (eight applicators and eight each environmentally exposed persons in Atlanta and Houston). In addition, the ages of the volunteers from whom hair samples were collected were not reported, and hair from older individuals could have higher accumulation of γ-HCH. Further, there was no information on whether any volunteers had previous exposure to γ-HCH applied to the scalp for treatment of lice.

A study conducted in Colorado indicated, in general, that no quantitative relationships were demonstrated between pesticide levels in household dust and pesticide levels in blood. However, γ-HCH levels in blood sera in a pesticide formulator (16.8 ng/g) and his wife (5 ng/g) were found to be elevated in a household in which dust levels measured 5.85 ng/g (Starr et al. 1974). It is possible that the γ-HCH found in the wife's blood and in the household came from the clothes and person of the pesticide formulator.

The Nonoccupational Pesticide Exposure Study (NOPES) conducted by EPA was based on the Total Exposure Assessment Methodology (TEAM) approach to exposure estimation. NOPES was designed to provide estimates of nonoccupational exposure to 32 household pesticides in the United States. Samples were collected at two locations: (1) Jacksonville, Florida, an area representative of high pesticide usage; and (2) Springfield/Chicopee, Massachusetts, an area of low-to-moderate pesticide usage. Detectable levels of γ-HCH were found in the personal air samples of 32–70% of the Jacksonville sample population; the range of mean concentrations in the air samples was 7–22 ng/m^3. For the Springfield population,

detectable levels of γ-HCH were found in personal air samples collected from 8–10% of the population, with mean concentrations of 0.7–5 ng/m^3 (EPA 1990).

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

The populations with the most potential for chronic exposure to HCH are workers who work at facilities that produce, process, or use γ-HCH. Exposure of the general population to γ-HCH tends to be low because federal regulations limiting its use have taken effect. However, γ-HCH is available in some prescription medications (e.g., shampoos, lotions), and the possibility of exposure may arise from use of these products. Individuals living near hazardous waste sites contaminated with HCH may also be exposed.

Historically, the largest occupational exposures came from people who work with pesticides. A study on occupational pesticide exposure of commercial seed-treating applicators was conducted in Montana (Grey et al. 1983). No exposure was detectable on the chest and arm pads, but γ-HCH was detected on the hands and on the respirator pads. Workers involved with γ-HCH application complained of nasal irritation if they did not wear a respirator or mask. The α-, β-, γ-, and δ-isomers of HCH have been detected in the blood serum and adipose tissue of individuals occupationally exposed to HCH in pesticide formulation. Serum levels of <0.5 ppb–1 ppm α-HCH, <0.9 ppb–0.72 ppm β-HCH, <0.7 ppb–0.17 ppm γ-HCH, and 0.002–0.16 ppm δ-HCH have been detected in exposed workers (Baumann et al. 1980; Kashyap 1986; Morgan and Lin 1978; Nigam et al. 1986). Mean adipose tissue levels of 5.8 mg α-HCH/kg, 45.6 mg β-HCH/kg, and 3.1 mg γ-HCH/kg have also been reported in exposed workers (Baumann et al. 1980).

A number of case reports (e.g., Bhalla and Thami 2004; Daud et al. 2010; Juan et al. 2004; Paul et al. 2013; Shah et al. 2013; Ramabhatta et al. 2014; Wiles et al. 2015; Yu et al. 2015) have documented toxic effects in humans overexposed to γ-HCH through excessive dermal application or accidental or intentional ingestion of products used to treat scabies and head lice; effects observed in these studies are described in Chapter 2.